Chapter 8

Wastewater Treatment and Waste Disposal

8. Wastewater Treatment and Waste Disposal

8.1. Introduction

Hydraulic fracturing used for the development of oil and gas resources results in the production of wastewater containing a range of problematic or potentially problematic constituents (see Chapter 7) and requiring management. For the purposes of this assessment, hydraulic fracturing wastewater encompasses flowback and produced water (often referred to together as produced water) that is managed using any of a number of practices, including treatment and discharge, reuse, or injection into Class IID wells regulated under the Underground Injection Control (UIC) program under the Safe Drinking Water Act (SDWA) (see also Chapter 1).[1] In this chapter, the term "wastewater" is generally used. In limited cases where more specific information is provided about a wastewater (e.g., a source indicates that the wastewater is flowback), that information is also noted.

Although wells producing from either unconventional or conventional oil and gas reservoirs generate produced water during the course of their productive lifespan, wells conducting modern high-volume hydraulic fracturing can generate a large volume of flowback water in the period immediately after fracturing. Stakeholders reported to the U.S. Government Accountability Office that flowback volumes could be 420,000 gal to 2.52 Mgal (10,000 to 60,000 bbl or 1.59 million to 9.54 million L) per well per hydraulic fracture (U.S. GAO, 2012) (see Chapter 7.1.1 for more information on produced water volumes per well in various geologic basins and plays). This necessitates having a wastewater management strategy in place at the beginning of activities at the well. Selection of management choices may depend upon the quality and volume of the fluids, logistics, and economics.

Treatment and disposal strategies vary throughout the United States and may include underground injection, on-site or offsite treatment for reuse in subsequent hydraulic fracturing operations, reuse without treatment, or other uses. In some areas, wastewater may be applied to the land (e.g., for irrigation) or held in pits for evaporation. The large majority of wastewater generated from all oil and gas operations in the United States is disposed of via Class IID wells (Clark and Veil, 2009). As hydraulic fracturing activity matures, costs of different disposal practices may change in various regions due to factors such as regulations, available infrastructure, feasibility and cost of reuse practices, and other concerns that are difficult to anticipate and quantify at the time of this assessment.

Over the past decade, the rapid increase in modern hydraulic fracturing activities has led to the need to manage the associated wastewater. There has been a shift towards reuse in areas where

[1] The term "wastewater" is being used in this study as a general description of certain waters and is not intended to constitute a term of art for legal or regulatory purposes. This general description does not, and is not intended to, provide that the production, recovery, or recycling of oil, including the production, recovery, or recycling of produced water or flowback water, constitutes "wastewater treatment" for the purposes of the Oil Pollution Prevention regulation (with the exception of dry gas operations), which includes the Spill Prevention, Control, and Countermeasure rule and the Facility Response Plan rule, 40 CFR 112 et seq.

there are relatively few Class IID wells (e.g., the Marcellus Shale region) and indications of interest in reuse in areas where access to water for fracturing is limited (e.g., Anadarko Basin in TX and OK). The term reuse is sometimes used to imply no treatment or basic treatment (e.g., media filtration) for the removal of constituents other than total dissolved solids (TDS), while recycling is sometimes used to convey more extensive treatment (e.g., reverse osmosis (RO)) to remove TDS (Slutz et al., 2012). In this document, the term "reuse" will be used to indicate use of wastewater for subsequent hydraulic fracturing, without regard to the level of treatment.

This chapter provides follow-on to Chapter 7, which discusses the composition and per-well volumes of produced water and the processes involved in its generation. In this chapter, discussions are included on management practices for hydraulic fracturing wastewaters, available wastewater production information, and estimated aggregate volumes of wastewater generated for several states with active hydraulic fracturing (Section 8.2). As a complement to information on the composition of wastewaters in Chapter 7, issues of concern associated with wastewater constituents are also presented (Section 8.3). Management methods that are used in 2014-2015 or have been used in recent years are described (Section 8.4). Information is then presented on the types and effectiveness of treatment processes that are suitable for removal of constituents of concern in hydraulic fracturing wastewaters, either in centralized waste treatment facilities (CWTs) or on-site treatment; examples of CWTs are presented (Section 8.5 and Appendix F). With the background provided in the earlier sections of the chapter, documented and potential impacts on drinking water resources are discussed (Section 8.6), and a final synthesis discussion is then provided (Section 8.7).

This chapter makes use of background information collected by the EPA's Office of Water (OW) as part of its development of proposed effluent limitations guidelines and standards for wastewater from unconventional oil and gas resources (U.S. EPA, 2015q). These are defined by guidelines and standards as resources in low permeability formations including oil and gas shales, tight oil, and low permeability sandstones and carbonates. Coalbed methane is beyond the scope of those standards. In this chapter we consider wastewater generated by hydraulic fracturing of those unconventional oil and gas resources included in the background research done by OW in addition to wastewater generated by hydraulic fracturing in coalbed methane plays and conventional reservoirs.

8.2. Volumes of Hydraulic Fracturing Wastewater

This section of Chapter 8 provides a general overview of aggregate wastewater quantities generated in the course of hydraulic fracturing and subsequent oil and gas production, including estimates at regional and state levels. It also discusses methodologies used to produce these estimates and the challenges associated with the preparation and use of available estimates. (Chapter 7 provides a more in-depth discussion of the processes affecting produced water volumes and presents some typical per-well values and temporal patterns.) Wastewater volumes most likely will vary in the future as the amount and locations of hydraulic fracturing activities change and as existing wells age and move into later phases of production. The volumes and management of wastewater are important factors affecting the potential for wastewater to affect drinking water resources.

The volume of wastewater generated is generally tied to the volume of oil and gas production; as operators increase hydrocarbon production, there will be a corresponding increase in wastewater volumes to be managed. For example, data from the Pennsylvania Department of Environmental Protection (PA DEP) (PA DEP, 2015a) (see Figure 8-1) show trends in volumes of wastewater compared to gas produced from wells in the Marcellus Shale in Pennsylvania. Although the data show some variation, they demonstrate a general correlation between wastewater and produced gas.

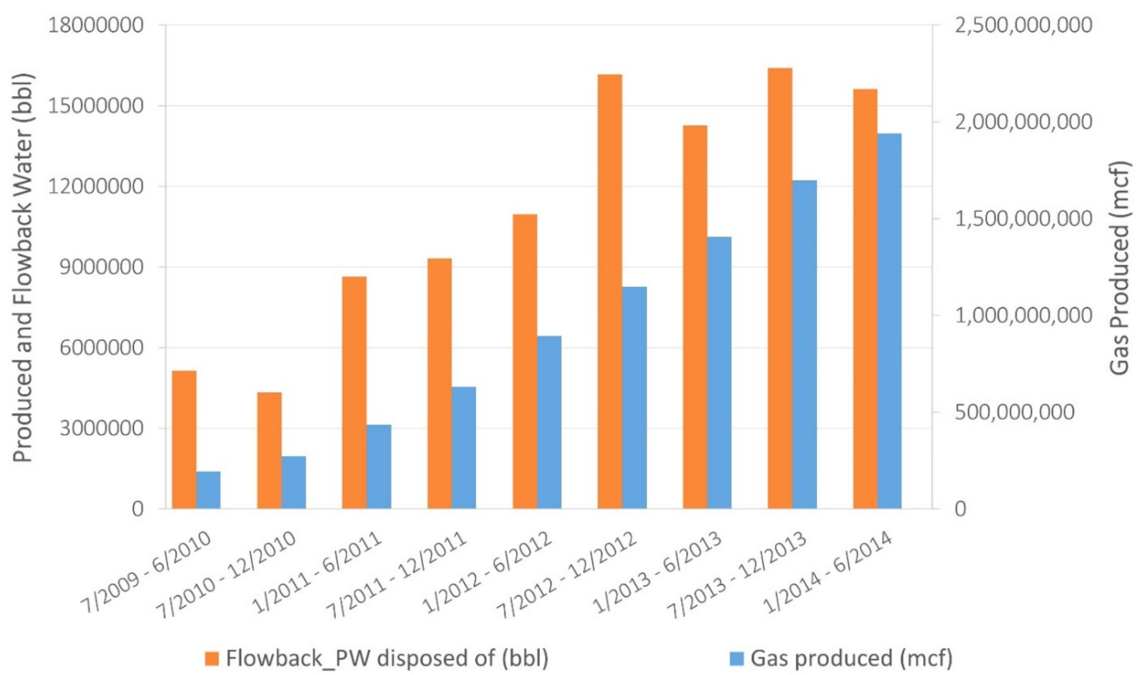

Figure 8-1. Produced and flowback water volumes and produced gas volumes from unconventional wells in Pennsylvania from July of 2009 through June of 2014.

Source: PA DEP (2015a).

Information presented in Chapter 7 highlights the initial rapid recovery of fluid in the first weeks after fracturing (see Figure 7-2), with a subsequent substantial reduction in the volume of water flowing through the well to the surface. This is followed by recovery of produced water during the longer-term productive phase of the well's life. One source suggests that, as a general rule of thumb, the amount of flowback produced in the days or weeks after hydraulic fracturing is roughly comparable to the amount of long-term produced water generated over a span of years, which may vary considerably among wells (IHS, 2013). Thus, on a local level, operators can anticipate a relatively large volume of wastewater in the weeks following fracturing, with slower subsequent production of wastewater. Wells also generate some amount of drilling-fluid waste. Compared to produced water, however, drilling wastewater can constitute a relatively small portion of the total wastewater produced (e.g., <10% in Pennsylvania during 2004-2013) (U.S. EPA, 2015q) and is not discussed further in this assessment.

In addition to variation in per-well wastewater volumes, aggregate wastewater production for an area or region will vary from year to year with hydraulic fracturing activity. For instance, the average annual volume of wastewater generated by all gas production (both shale gas and conventional) in Pennsylvania quadrupled from the 2001-2006 period to the 2008-2011 period. During the latter period, wastewater volume averaged 1.1 billion gal (26 million bbl or 4.2 billion L) per year (Wilson and Vanbriesen, 2012).

8.2.1. National Level Estimate

Clark and Veil (2009) estimated that in 2007, the approximately one million active oil and gas wells in the United States generated approximately 2.4 billion GPD (57.4 million bbl/day; 9.1 billion L/day) of wastewater; no newer comprehensive national-level estimate exists in the literature as of April 2015. Note that this estimate is not limited to wastewater from hydraulic fracturing operations. This national-level estimate is reported to represent total oil and gas wastewater (from conventional and unconventional resources, and from wells hydraulically fractured and wells not hydraulically fractured), but the authors note that it does not include the flowback water component. Although Clark and Veil (2009) conducted a state-by-state analysis, the report may have underestimated production due to significant data limitations: 1) data based on a timeframe preceding the dramatic increase in hydraulic fracturing activity seen in more recent years; 2) estimates based on data that were collected and maintained in a variety of ways, making data synthesis difficult; and 3) incomplete data (U.S. GAO, 2012).

8.2.2. Regional/State and Formation Level Estimates

The amount of wastewater generated in a given region varies widely depending upon the volume of wastewater generated per well and the number of wells in the area. The factors influencing wastewater production are discussed in Chapter 7, which also discusses differences among formations in volumes recovered during flowback and long-term water production. Table 7-2 provides rates of produced water generation for a number of formations in the United States.

At an aggregate level, wastewater volumes and geographic variability have been assessed for oil and gas production in several studies. A 2011 study by the Bureau of Land Management (BLM) (Guerra et al., 2011) states that more than 80% of oil and gas wastewater is generated in the western United States, including volumes from both conventional and unconventional resources. The BLM report notes substantial contributions from coalbed methane (CBM) wells, in particular those in the Powder River Basin (Wyoming). The authors state that Wyoming produces the second highest volume of water among the western United States. Guerra et al. (2011) also highlight the large portion of wells and wastewater associated with Texas (44% of U.S. produced water volume). Although the authors do not identify all wastewater contributions from production involving hydraulic fracturing, the regions with established oil and gas production are likely to have methods and infrastructure available for management of hydraulic fracturing wastewater. Figure 8-2 summarizes the findings for these western states, demonstrating the wide variability in wastewater from state to state (likely reflecting differences in formation geology and oil and gas production activity).

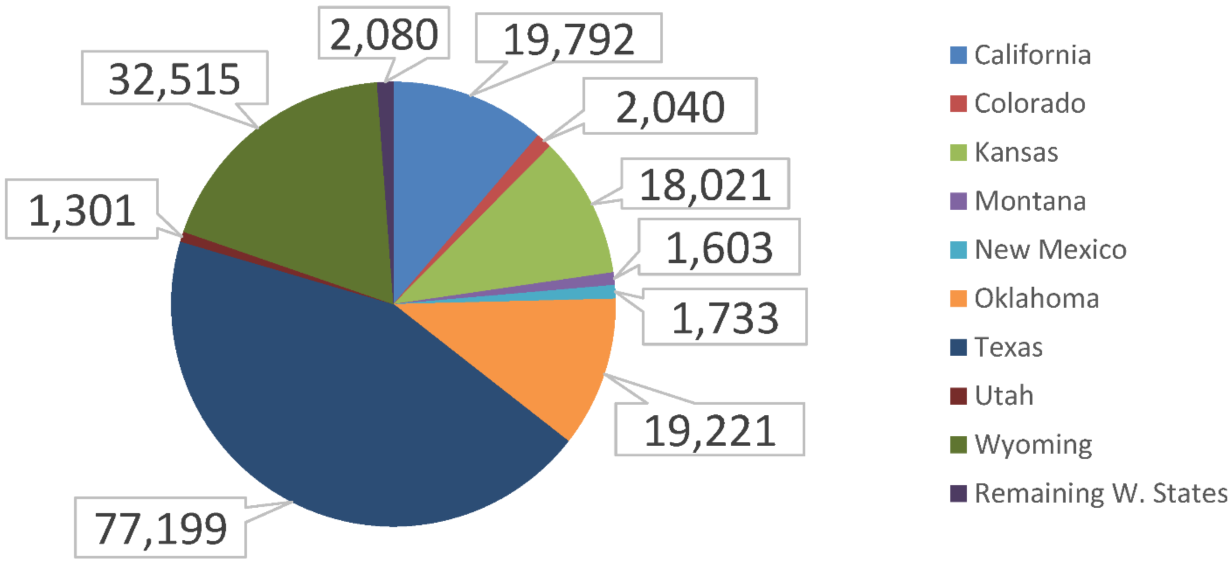

Figure 8-2. Wastewater quantities in the western United States (billions of gallons per year).

Source: Guerra et al. (2011).

Table 8-1 presents estimates of the numbers of wells and volumes of hydraulic fracturing wastewater generated in North Dakota, Ohio, Pennsylvania, Texas, and West Virginia. The data shown in this table come from publicly available state websites and databases; data for West Virginia reference a report by Hansen et al. (2013) that compiled available flowback data from West Virginia. The reported volumes have been summed by year. These states are represented in Table 8-1 because the produced water volumes were readily identifiable as associated with hydraulic fracturing activity. Differences in the years presented for the states are due to differences in data availability from the state agency databases. However, the increases in the numbers of wells producing wastewater and the volumes of wastewater produced are generally consistent with the timing of the expansion of high-volume hydraulic fracturing and track with the increase in horizontal wells seen in Figure 2-12.

Several states with mature oil and gas industries (California, Colorado, New Mexico Utah, Wyoming) make produced water volumes publicly available by well as part of their oil and gas production data, but they do not directly indicate which wells have been hydraulically fractured. Some states (Colorado, Utah, and Wyoming) specify the producing formation along with volumes of hydrocarbons and produced water. New Mexico provides data for separate basins as well as for the entire state. Determining volumes of hydraulic fracturing wastewater for these states is challenging because there is a possibility of either inadvertently including wastewater from wells not hydraulically fractured or of missing volumes that should be included. Appendix Table F-1 provides estimates of wastewater volumes in these states in regions where hydraulic fracturing activity is taking place along with notes on data limitations.

The data in Table 8-1 and Appendix Table F-1 illustrate the challenges both for compiling a national estimate of hydraulic fracturing wastewater as well as comparing wastewater production among states due to dissimilar data types, presentation, and availability.

Table 8-1. Estimated volumes (millions of gallons) of wastewater based on state data for selected years and numbers of wells producing fluid.

State	Basin	Principal Lithologies	Data Type	2000	2004	2008	2010	2011	2012	2013	2014	Comments
North Dakota	Williston	Shale	Produced water	2	3	130	790	1,900	4,500	8,500	9,700	From North Dakota Oil and Gas Commission website.[a] Data provided for six members of the Bakken Shale. Produced water includes flowback (reports are submitted within 30 days of well completion.)
			Wells	*161*	*152*	*844*	*2,083*	*3,303*	*5,036*	*6,913*	*8,039*	
Ohio	Appalachian	Shale	Primarily flowback water	-	-	-	-	3	29	110	-	Data from Ohio DNR Division of Oil and Gas.[b] Utica data for 2011 and 2012. Utica and Marcellus data for 2013. Brine is noted by agency to be mostly flowback.
			Wells	-	-	-	-	*9*	*86*	*400*	-	
Pennsylvania	Appalachian	Shale	Flowback water	-	-	-	92	340	410	350	210	Waste data from PA DEP.[c] 2nd half of 2010 and first half of 2014. Data described as unconventional as determined by formation. Separate codes are provided by PA DEP for flowback and produced water.
			Wells	-	-	-	*334*	*1,564*	*1,622*	*1,465*	*895*	
			Produced water	-	-	-	90	400	730	930	440	
			Wells	-	-	-	*1, 035*	*1,826*	*3,665*	*4,761*	*4,889*	

State	Basin	Principal Lithologies	Data Type	2000	2004	2008	2010	2011	2012	2013	2014	Comments
Pennsylvania cont.	Appalachian cont.	Shale, cont.	Flowback and produced water	-	-	-	180	740	1,100	1,300	650	
			Wells	-	-	-	*1,232*	*2,434*	*4,039*	*5,015*	*5,150*	
Texas	Unspecified (entire state)	Shale, Sandstone	Flowback water - injected volumes	-	-	-	-	490	2,200	3,100	2,000	Waste injection data from Texas Railroad Commission.[d] Monthly totals are provided for entire state. Oct - Dec for 2011, full years for 2012 and 2013, and Jan - Oct for 2014
West Virginia	Appalachian	Shale	Flowback water - Estimated total disposed	-	-	-	120	110	59	-	-	Estimates from Hansen et al. (2013).

[a] North Dakota Industrial Commission. Department of Mineral Resources. Bakken Horizontal Wells By Producing Zone: https://www.dmr.nd.gov/oilgas/bakkenwells.asp.

[b] Ohio Department of Natural Resources, Division of Oil and Gas Resources. Oil and Gas Well Production. http://oilandgas.ohiodnr.gov/production#ARCH1.

[c] Pennsylvania Department of Environmental Protection. PA DEP Oil and Gas Reporting Web site. https://www.paoilandgasreporting.state.pa.us/publicreports/Modules/Welcome/Agreement.aspx

[d] Railroad Commission of Texas. Injection Volume Query. http://webapps.rrc.state.tx.us/H10/searchVolume.do;jsessionid=J3cgVHhK9nkwPrC7ZcWNMgyzF9LCYyR1NmvDy3F1QQ5wqXfcGNGN!1841197795?fromMain=yes&sessionId=143075601021612. Texas state data provide an aggregate total amount of flowback fluid injected for the past few years. (Data on brine volumes injected do not differentiate hydraulically fractured wells and are therefore not presented here.) These values are interpreted as estimates of generated flowback water as based on reported quantities of “fracture water flow back” injected into Class IID wells.

8.2.3. Estimation Methodologies and Challenges

Compiling and comparing data regarding wastewater production at the wide array of oil and gas locations in the United States presents challenges, and various approaches are used to estimate wastewater volumes, both at the state and national level. Data from state agency web sites and databases can be a ready source of information, whether publicly available and downloadable or provided directly by agencies upon request. However, due to sometimes significant differences in the types of data collected, mechanisms, formats, and definitions used, data cannot always be directly compared from state to state and can be difficult to aggregate at a national level. The inconsistences encountered in data searches for this assessment agree with recent conclusions by Malone et al. (2015), who noted inconsistences among 10 states with unconventional oil and gas activity in the accessibility, usability, completeness, accuracy, and cost of various types of data (e.g., wells drilled, production, waste, Class IID wells).

One challenge associated with using state production data to estimate the volume of wastewater nationally or regionally is the lack of consistency in data collection (U.S. GAO, 2012). Some states do not include a listing of wastewater (usually listed as produced water volumes) in their publicly available oil and gas production reports, while others do. State tracking of wastewater volumes may or may not include information that helps in determining whether the producing well was hydraulically fractured (e.g., an indicator of resource type or formation). It also might not be clear whether volumes listed as produced water include the flowback component. Some states (e.g., Colorado) include information on disposal and management methods along with production data, and others do not.

Given these limitations, some studies have generated estimates of wastewater volume using water-to-gas and water-to-oil ratios along with the reports of hydrocarbon production (Murray, 2013). The reliability of any wastewater estimates made using this method will need to be evaluated in terms of the quality, timeframe, and spatial coverage of the available data, as well as the extent of the area to which the estimates will be applied. Water-to-hydrocarbon ratios are empirical estimates. Because these ratios show a wide variation among formations, reliable data are needed to formulate a ratio in a particular region.

Another approach to estimating wastewater volumes would entail multiplying per-well estimates of flowback and produced water production rates by the numbers of wells in a given area. Challenges associated with this approach include obtaining accurate estimates of the number of new and existing wells, along with accurate estimates of per-well water production both during the flowback period and during the production phase of the wells' lifecycle. In particular, it can be challenging to correctly match per-well wastewater production estimates, which will vary by formation, with counts of wells, which may or may not be clearly labeled by or associated with specific formations. Temporal variability in wastewater generation would also be difficult to capture and would add to uncertainty. Such an approach, however, may be attempted for order of magnitude estimates if the necessary data are available and reliable.

8.3. Wastewater Characteristics

Along with wastewater volume, wastewater characteristics are important for understanding the potential impacts of management and disposal of hydraulic fracturing wastewater on drinking water resources. Chapter 7 provides in-depth detail on produced water chemistry. This section provides brief highlights of the important features of wastewater composition as well as the characteristics of the residuals produced during wastewater treatment.

8.3.1. Wastewater

This section briefly discusses why the composition of hydraulic fracturing wastewaters needs to be considered when planning for wastewater management, especially if treatment or reuse are planned. Concerns associated with selected constituents are presented; treatment considerations associated with various wastewater constituents are included in Section 8.5.

8.3.1.1. Total Dissolved Solids and Inorganics

Wastewaters are generally high in total dissolved solids (TDS), especially waters from shale and tight sandstone formations, with values ranging from less than 1,000 mg/L to hundreds of thousands of mg/L (see Section 7.6.4 and Table 7-4). The TDS in wastewaters from shale formations is typically dominated by sodium and chloride and may also include elevated concentrations of bromide, bicarbonate, sulfate, calcium, magnesium, barium, strontium, radium, organics, and heavy metals (Chapman et al., 2012; Rowan et al., 2011; Blauch et al., 2009; Orem et al., 2007; Sirivedhin and Dallbauman, 2004). Within each play, the minimum and maximum values shown in Table 7-4 suggest spatial variation that may need to be accommodated when considering management strategies such as reuse or treatment. In contrast to shales and sandstones, TDS values for wastewater from CBM formations are generally lower, with concentrations ranging from approximately 250 mg/L to 39,000 mg/L (Benko and Drewes, 2008; Van Voast, 2003) (see Appendix Table E-3). This results in fewer treatment challenges and a wider array of management options.

Although TDS has a secondary maximum contaminant level (MCL) (secondary MCLs are non-mandatory water quality standards) of 500 mg/L for aesthetic purposes, it is not considered a health-based contaminant and is therefore not regulated under the EPA's National Primary Drinking Water Regulations, although other standards may apply. For example, a maximum concentration of 500 mg/L has been used by the state of Pennsylvania for some industrial wastewater discharges. Constituents commonly found in TDS from hydraulic fracturing wastewaters may have potential impacts on health or create burdens on downstream drinking water treatment plants if discharged at high concentrations to drinking water resources. Bromide, for example, can contribute to the increased formation of disinfection by-products (DBPs) during drinking water treatment (Hammer and VanBriesen, 2012); see Section 8.6.1.

Metals (e.g., barium, cadmium, chromium, lead, copper, manganese, nickel, thallium, and zinc) present in TDS can be toxic to humans and aquatic life at certain concentrations. Health effects of these metals can include kidney damage, liver damage, skin conditions, high blood pressure, and developmental problems (U.S. EPA, 2015i). To ensure safe drinking water, the EPA has established primary MCLs for a number of these constituents. MCLs and action levels for these metals vary from 0.002 mg/L for thallium to 1.3 mg/L for copper (U.S. EPA, 2015i). Cadmium has been found in produced water from tight gas formations at concentrations as high as 0.37 mg/L (the MCL is 0.005 mg/L), and chromium has been found at concentrations up to 0.265 mg/L (the MCL is 0.1 mg/L) (see Table 7-4).

Other constituents of concern among dissolved solids are chloride, sulfate, barium, and boron. Elevated concentrations of chloride and sulfate are of concern because of drinking water aesthetics, and the EPA has established secondary MCLs for both chloride and sulfate of 250 mg/L (U.S. EPA, 2015i; Hammer and VanBriesen, 2012). Barium has a primary MCL of 2 mg/L and has been found in some shale gas produced waters at concentrations in the thousands of mg/L (see Table 7-4). Boron is not regulated under the National Primary Drinking Water Regulations, but internal plant specifications for one CWT (e.g., the Pinedale Anticline Facility) and waste discharge requirements (WDR) permit for another (e.g., San Ardo Water Reclamation Facility) limit boron effluent concentrations to 0.75 mg/L (Shafer, 2011; Webb et al., 2009).

8.3.1.2. Organics

Less information is available about organic constituents in hydraulic fracturing wastewaters than about inorganic constituents, but there are several studies that include some analyses of organic constituents. The organic content in flowback waters can vary based on the chemical additives used and the formation but generally consists of polymers, oil and grease, volatile organic compounds (VOCs), and semi-volatile organic compounds (SVOCs) (Walsh, 2013; Hayes, 2009). Examples of other constituents detected include alcohols, naphthalene, acetone, and carbon disulfide (U.S. EPA, 2015i) (see Appendix Table E-10). Wastewater associated with CBM wells may have high concentrations of aromatic and halogenated organic contaminants that that may require treatment depending on how the wastewater will be managed or disposed of (Pashin et al., 2014; Sirivedhin and Dallbauman, 2004). Concentrations of BTEX (benzene, toluene, ethylbenzene, and xylenes), in CBM produced waters are, however, lower than in shale produced waters (see Appendix Table E-9).

Certain organic compounds are of concern in drinking water because they can cause damage to the nervous system, kidneys, and/or liver and can increase the risk of cancer if ingested over a period of time (U.S. EPA, 2006). Some organics in chemical additives are known carcinogens, including 2-butoxyethanol (2BE), naphthalene, benzene, and polyacrylamide (Hammer and VanBriesen, 2012). Many organics are regulated for drinking water under the National Primary Drinking Water Regulations. Section 8.6.4 provides further discussion of documented or potential situations in which organic constituents have or might reach drinking water resources.

8.3.1.3. Radionuclides

Radionuclides are constituents of concern in some hydraulic fracturing wastewaters, with most available data obtained for the Marcellus Shale in Pennsylvania (see Appendix Table E-8). Results from a USGS report (Rowan et al., 2011) indicate that radium-226 and radium-228 are the predominant radionuclides in Marcellus Shale wastewater, and they account for most of the gross alpha and gross beta activity in the waters studied. There are limited data on radionuclides in wastewater from formations other than the Marcellus Shale, but information on the naturally occurring radioactive material (NORM) in the formations themselves, in particular uranium and thorium, may suggest the potential for high levels of radionuclides in produced water, especially where TDS concentrations are also high. Sections 7.5.4 and 7.6.6 provide further information on radionuclides in formations and in produced waters.

The primary radioactive contaminants found in hydraulic fracturing wastewaters (radium, gross alpha radiation, and gross beta radiation) can increase the risk of cancer if consumed at elevated levels over time (U.S. EPA, 2015i). Therefore, the EPA has established drinking water MCLs for combined radium (radium-226 plus radium-228), gross alpha, and gross beta of 5 pCi/L, 15 pCi/L, and 4 millirems/year, respectively (see Section 8.6.2).

8.3.2. Constituents in Residuals

Depending on the water being treated and treatment processes used, treatment residuals may consist of sludges, spent media (used filter materials), or brines. Residuals can include constituents such as total suspended solids (TSS), TDS, metals, radionuclides, and organics. The treatment process tends to concentrate wastewater constituents in the residuals. As an illustration of the degree of concentration that can take place, processes such as electrodialysis and mechanical vapor recompression have been found to yield residuals streams with TDS concentrations in excess of 150,000 mg/L, from treating waters with influent TDS concentrations of approximately 50,000 – 70,000 mg/L (Hayes et al., 2014; Peraki and Ghazanfari, 2014).

Also, technologically enhanced naturally occurring radioactive material (TENORM) in wastewaters may cause residual wastes to have elevated gamma radiation emissions (Kappel et al., 2013).[1] One study calculated that typical solids produced by precipitation processes designed to remove barium and strontium from Marcellus Shale wastewater would contain between 2,571 and 18,087 pCi/g of radium in the barium sulfate precipitate (Zhang et al., 2014b). Another similar study using mass balances calculated that sludge from a sulfate precipitation process would average a radium concentration of 213 pCi/g in sludge (Silva et al., 2012). Silva et al. (2012) estimated a radium-226 concentration of 58 pCi/g in sludge from lime softening processes, a level that would necessitate disposal of low level radioactive waste.

8.4. Wastewater Management Practices

Operators have several strategies for management of hydraulic fracturing wastewaters (see Figure 8-3), with the most common choice being disposal via Class IID wells (Clark et al., 2013; Hammer and VanBriesen, 2012). Other practices include reuse in subsequent hydraulic fracturing operations (with varying levels of treatment), treatment at a CWT (often followed by reuse), evaporation (in arid regions), or in some cases, depending on state and local requirements, various other wastewater management strategies (e.g., irrigation, which involves no discharge to waters of the U.S.). The management methods shown in Figure 8-3 represent various strategies, not all of which will happen together.

[1] Technologically enhanced naturally occurring radioactive materials (TENORM) are radionuclides that have been concentrated or enhanced as the result of human activity.

At one time, treatment of unconventional oil and gas wastewaters at publicly owned treatment works (POTWs) was a common practice for wastewater management in the Marcellus region (Lutz et al., 2013). However, this practice has been essentially discontinued following a request from PA DEP that, by May 19, 2011, oil and gas operators stop sending Marcellus Shale wastewater to 15 POTWs and CWTs that discharged to surface waters (U.S. EPA, 2015h).

Each of these wastewater management strategies may potentially lead to an impact on drinking water resources during some phase of their execution. Such impacts may include accidental releases during transport (see Chapter 7), discharges of treated wastewaters from CWTs or POTWs where treatment for certain constituents has been inadequate, migration of constituents in wastewaters that have been applied to land, leakage from on-site storage pits (see Chapter 7), inappropriate management of residuals (e.g., leaching from landfills or land application), or accumulation of constituents in sediments near outfalls of CWTs or POTWs that have treated hydraulic fracturing wastewater.

A reliable census of nationwide wastewater management practices is difficult to assemble due to a lack of consistent and comparable data among states, but Table 8-2 illustrates the variability in the primary wastewater management methods using available qualitative and quantitative sources. Disposal via underground injection predominates in most regions. Reuse is most prevalent in the Appalachian Basin in Pennsylvania. Moderate reuse occurs in the Arkoma (OK, AR) and Anadarko (OK, TX) basins, and use of CWTs occurs predominantly in Pennsylvania.

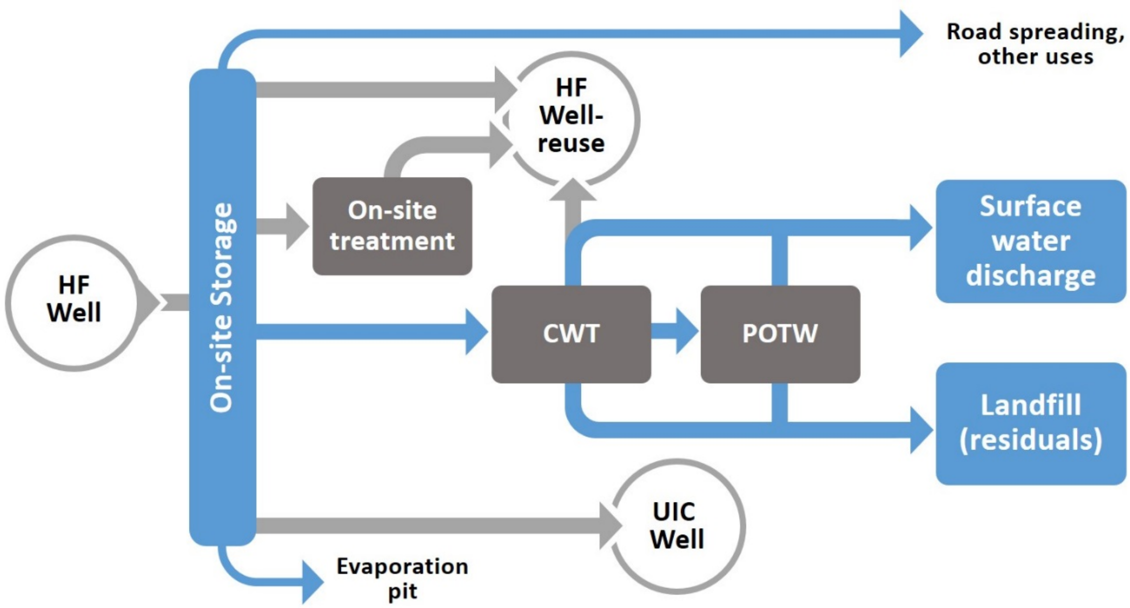

Figure 8-3. Schematic of wastewater management strategies.

Table 8-2. Hydraulic fracturing wastewater management practices in recent years.

Source: (U.S. EPA, 2015q).

Basin	Formation	Resource type	Reuse	Injection for disposal	CWT facilities	Notes	Available data[b]
Michigan	Antrim	Shale Gas		XXX			Qualitative
Appalachian	Marcellus/Utica (PA)	Shale Gas	XXX	XX	XX	Limited Class IID wells in east	Quantitative
	Marcellus/Utica (WV)	Shale Gas/Oil	XXX	XX	X		Quantitative
	Marcellus/Utica (OH)	Shale Gas/Oil	XX	XXX	X		Mixed
Anadarko	Granite Wash	Tight Gas	XX	XXX	X[a]		Mixed
	Mississippi Lime	Tight Oil	X	XXX		Reuse limited but is being evaluated	Qualitative
	Woodford; Cana; Caney	Shale Gas/Oil	X	XXX	X[a]		Qualitative
Arkoma	Fayetteville	Shale Gas	XX	XX	X[a]	Few existing Class IID wells; new CWT facilities are under construction	Mixed
Fort Worth	Barnett	Shale Gas	X	XXX	X[a]	Reuse not typically effective due to high TDS early in flowback and abundance of Class IID wells	Mixed
Permian	Avalon/Bone Springs, Wolfcamp, Spraberry	Shale/tight Oil/gas	X	XXX	X[a]		Mixed
TX-LA-MS Salt	Haynesville	Tight Gas	X	XXX		Reuse not typically cost effective due to high TDS early in flowback and abundance of Class IID wells	Mixed

Basin	Formation	Resource type	Reuse	Injection for disposal	CWT facilities	Notes	Available data[b]
West Gulf	Eagle Ford, Pearsall	Shale Gas/Oil	X	XXX	X		Mixed
Denver Julesburg	Niobrara	Shale Gas/Oil	X	XXX	X		Mixed
Piceance; Green River	Mesaverde/Lance	Tight Gas	X	XX	X	Also managed through evaporation to atmosphere in ponds in this region	Qualitative
Williston	Bakken	Shale Oil	X	XXX		Reuse limited but is being evaluated	Mixed

[a] CWT facilities in these formations are operator owned.

[b] This column indicates the type of data on which EPA based the number of X's. In most cases, EPA used a mixture of qualitative and quantitative data sources along with engineering judgment to determine the number of X's.

XXX—The majority (≥ 50%) of wastewater is managed with this management practice.

XX—A moderate portion (≥ 10% and < 50%) of wastewater is managed with this management practice.

X—This management practice has been documented in this location but for a small (< 10%) or unknown percent of wastewater.

Management choices are affected by cost and a number of other factors, including the chemical properties of the wastewater; the volume, duration, and flow rate of the water generated; the logistical feasibility of various options; the availability of necessary infrastructure; federal, state, and local regulations; and operator discretion (U.S. GAO, 2012; NPC, 2011a). The economics (such as transport, storage, and disposal costs) and availability of various treatment and disposal methods are of primary importance (U.S. GAO, 2012). For example, as of early 2015, Pennsylvania has nine operating Class IID wells within the state, whereas Texas has nearly 7,900 (U.S. EPA, 2015q).

The availability and use of management strategies may change in a region over time as oil and gas development increases or decreases, changing the volumes of wastewater that need to be handled on a local, state, and regional level (see Text Box 8-1 for more information on hydraulic fracturing wastewater management in Pennsylvania). Figure 8-4 illustrates shifting wastewater management practices in Pennsylvania over the last several years as shale gas development has proceeded in the Marcellus Shale. On-site reuse (labeled as "Reuse HF" in Figure 8-4) has grown. Also, most CWT management of Marcellus wastewater in recent years has been at zero-discharge facilities (i.e., for reuse) (an estimated 80% in 2012 and 90% in 2013) (PA DEP, 2015a). Combined with the volumes managed via on-site reuse, Pennsylvania reuse rates are approximately 85% to 80%. In contrast, wastewater disposal data for Colorado (see Figure 8-5) show a steady use of injection wells (injected on lease) since 2000, and an apparent decrease in the use of onsite pits (state data were filtered for formations indicated in the literature to be targets for hydraulic fracturing).

Text Box 8-1. Temporal Trends in Wastewater Management – Experience of Pennsylvania.

Gross natural gas withdrawals from shale formations in the United States increased 518% between 2007 and 2012 (EIA, 2014c). This production increase has led to larger volumes of wastewater that require appropriate management (Vidic et al., 2013; Gregory et al., 2011; Kargbo et al., 2010). The rapid increase in wastewater generated from oil and gas wells used for hydraulic fracturing has led to many changes in the wastewater disposal practices in the oil and gas industry. Changes have been most evident in Pennsylvania, which has experienced more than a 1,400% increase in natural gas production since 2000 (EIA, 2014c).

Lutz et al. (2013) estimated that total wastewater generation in the Marcellus region increased 570% between 2004 and 2013 and concluded that this increase has created stress on the existing wastewater disposal infrastructure. In 2010, in response to concerns over elevated TDS in the Monongahela River basin and studies linking high TDS (and in particular high bromide levels) to elevated DBP levels in drinking water systems (PA DEP, 2011a), PA DEP amended Chapter 95 Wastewater Treatment Requirements under the Clean Streams Law for new discharges of TDS in wastewaters. This regulation is also known informally as the 2010 TDS regulation. The regulation disallowed any new indirect discharges (i.e., discharges to POTWs) of hydraulic fracturing waste and set limits of treated discharges from CWTs of 500 mg/L TDS, 250 mg/L chloride, 10 mg/L barium, and 10 mg/L strontium. Existing discharges were exempt.

In April 2011, PA DEP requested that oil and gas well operators transporting unconventional wastewater to the eight CWTs and seven POTWs that were exempt from the 2010 TDS regulation voluntarily stop discharging to these facilities. Follow-up letters from PA DEP to the owners of the wells specified that the role of bromides from Marcellus Shale wastewaters in the formation of total trihalomethanes (TTHM) was of concern (PA DEP, 2011a).

Between early 2011 and late 2011, although reported wastewater flows more than doubled, Marcellus drilling companies in Pennsylvania reduced their wastewater flows to CWTs that were exempt from the 2010 TDS regulation by 98%, and discharge to POTWs was 'virtually eliminated' (Hammer and VanBriesen, 2012).

Along with the decreased discharges from POTWs, there has been increased reuse of wastewater in the Marcellus Shale region. From 2008-2011, reuse of Marcellus wastewater has increased, POTW treatment volumes have decreased, tracking of wastewater has improved, and wastewater transportation distances have decreased (Rahm et al., 2013). Maloney and Yoxtheimer (2012) analyzed data from 2011 and found that reuse of flowback water increased to 90% by volume. Disposed flowback water comprised 8% of the total volume. Brine water, which was defined as formation water, was reused (58%), disposed via injection well (27%), or sent to industrial waste treatment plants (14%). Of all the fluid wastes in the analysis, brine water was most likely to be transported to other states (28%). They also concluded that wastewater disposal to municipal sewage treatment plants declined nearly 100% from the first half of 2011 to the second half.

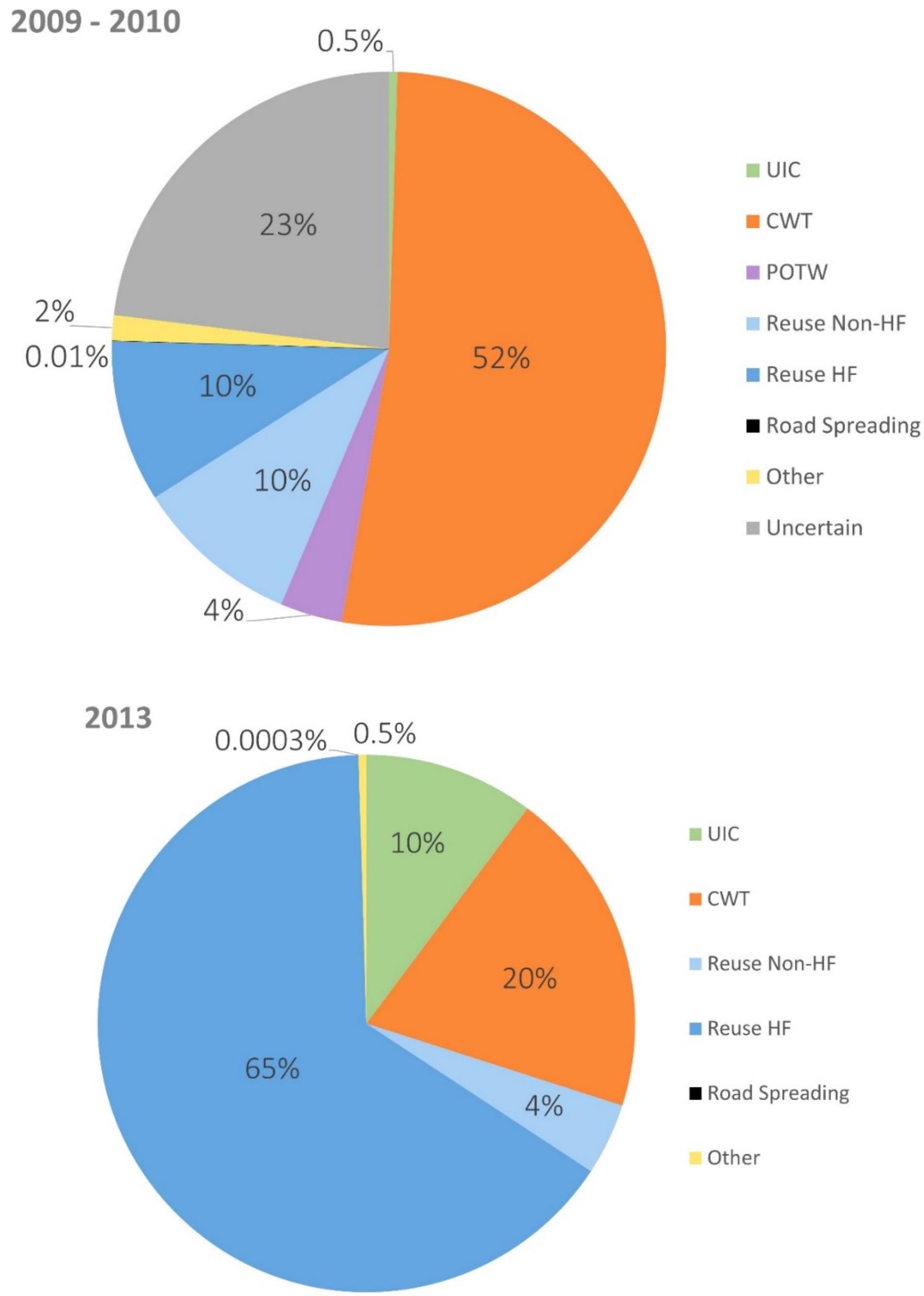

Figure 8-4. Percentages of Marcellus Shale wastewater managed via various practices for (top) the second half of 2009 and first half of 2010 (total estimated volume of 216 Mgal), and (bottom) 2013 (total estimated volume of 1.3 billion gallons).

"Reuse HF" indicates on-site reuse. Source: Waste data from PA DEP (2015a).

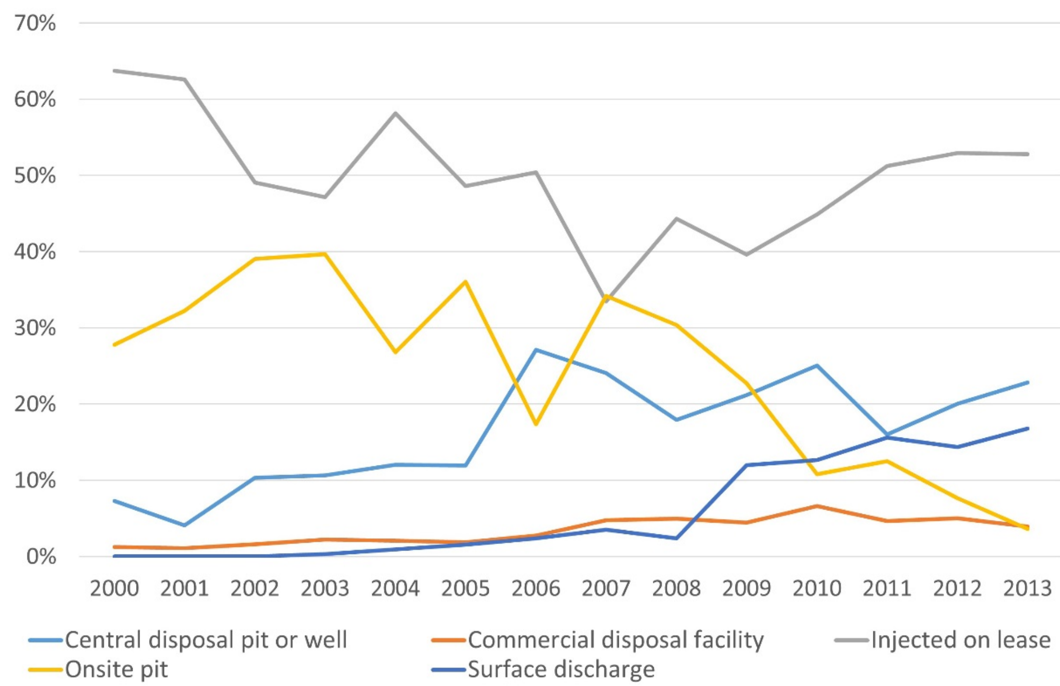

Figure 8-5. Management of wastewater in Colorado in regions where hydraulic fracturing is being performed.

Source: Production data from COGCC (2015).

Regulations also affect management options and vary geographically. At the Federal level, existing oil and gas effluent limitations guidelines and standards (ELGs) can be found under 40 CFR Part 435. These ELGs apply to conventional and unconventional oil and gas extraction facilities in various subcategories (e.g., Offshore, Onshore, Stripper Wells), with the exception of CBM discharges, which are not subject to the existing regulations. Subpart C, the Onshore subcategory, prohibits the discharge of wastewater pollutants to waters of the U.S. from onshore oil and gas extraction facilities. This "zero-discharge standard" means that oil and gas produced water pollutants cannot be directly discharged to surface waters. Operators have met these regulations through underground injection, reuse, or transfer of produced water to POTWs and/or CWTs. West of the 98th meridian (the arid western portion of the continental United States), discharges of wastewater from onshore oil and gas extraction facilities may be permitted for direct discharge to waters of the U.S. if the produced water has a use in agriculture or wildlife propagation when discharged into navigable waters. Definitions in 40 CFR 435.51(c) explain that the term "use in agricultural or wildlife propagation" means that (1) the produced water is of good enough quality to be used for wildlife or livestock watering or other agricultural uses; and (2) the produced water is actually put to such use during periods of discharge. The regulations at 40 CFR 435.52 specify that the only allowable discharge is produced water, with an oil and grease concentration not exceeding 35 milligrams per liter (mg/L). The regulations prohibit the discharge of waste pollutants into navigable waters from any source (other than produced water) associated with production, field exploration, drilling, well completion, or well treatment (i.e., drilling muds, drill cuttings, produced sands).

Unpermitted discharges of wastes related to hydraulic fracturing have been described in a number of instances. In Pennsylvania, discharges of brine into a storm drain that discharges to a tributary of the Mahoning River in Ohio. Analyses of the brine and drill cuttings that were discharged indicated the presence of contaminants, including benzene and toluene (U.S. Department of Justice, 2014). In California, an oil production company periodically discharged hydraulic fracturing wastewaters to an unlined sump for 12 days. It was concluded by the prosecution that the discharge posed a threat to groundwater quality (Bacher, 2013). These unauthorized discharges represent both documented and potential impacts on drinking water resources. However, data do not exist to evaluate whether such episodes are uncommon or whether they happen on a more frequent basis and remain largely undetected.

The following section provides an overview of hydraulic fracturing wastewater management methods, with some discussion of the geographic and temporal variations in practices. Discussion is provided on common treatment and disposal methods including on-site storage, underground injection, CWTs, reuse of hydraulic fracturing fluids, and evaporation methods. This section also provides discussion on past treatment of hydraulic fracturing wastewater at POTWs. Other management practices are also covered. Brief descriptions of treatment technologies applicable to hydraulic fracturing wastewater are available in Appendix F.

8.4.1. Underground Injection

Oil and gas wastewater may be disposed of via Class II injection wells regulated under the Underground Injection Control (UIC) Program under the Safe Drinking Water Act (SDWA)[1]. Class II wells include those used for enhanced oil recovery (IIR), disposal (IID), and hydrocarbon storage (IIH). Nationwide, injection wells dispose of a large fraction of wastewater from the oil and gas industry, including wastewater associated with hydraulic fracturing. A 2009 study notes that the oil and gas industry in the United States generated about 882 billion gal (21 billion bbl or 3.34 trillion L) of produced water in 2007 (Clark and Veil, 2009). More than 98% of this volume was managed via some form of underground injection, with 40% injected into Class II wells. However, a good national estimate of the amount of hydraulic fracturing wastewater injected into Class II wells is difficult to develop due to lack of available on data injection volumes specific to hydraulic fracturing operations that are compiled and able to be compared among states. Also, wastewater management methods are not well tracked in all states. Regional numbers of Class IID wells and generally low reuse rates (see Section 8.4.3), however, are consistent with Class IID wells being a primary means of wastewater management in many areas with hydraulic fracturing activity.

This assessment does not address whether there are documented or potential impacts on drinking water resources associated with the injection of hydraulic fracturing wastewaters into Class IID wells. However, should the feasibility of managing hydraulic fracturing wastewater via

[1] States may be given federal approval to run a UIC program under Section 1422 or 1425 of SDWA.

underground injection be limited in any way or become less economically advantageous, operators will likely adjust their wastewater management programs to favor other local practices such as treatment and discharge or reuse. Any new wastewater management decisions would then have to be evaluated in terms of potential impacts on drinking water resources.

The decision to inject hydraulic fracturing wastewater into Class IID wells depends, in part, on cost and on the proximity of the production well to the disposal well (and, therefore, transportation costs). For oil and gas producers, underground injection is usually the least expensive management strategy unless significant trucking is needed to transport the wastewater to a disposal well (U.S. GAO, 2012).

Class IID wells are not distributed uniformly among states due to differences in geology (including depth and permeability of formations), permitting, and historical demand for disposal of oil and gas wastewater. Table 8-3 shows the numbers of active Class IID wells across the United States, with the total count at a little over 27,000. The greatest numbers of wells are found in Texas, Oklahoma, and Kansas. For example, Texas has nearly 7,900 Class IID wells, with an estimated daily disposal volume of approximately 400 million gal per day (MGD) (1.5 billion L/day) (see Table 8-3). This large disposal capacity in Texas is consistent with the availability of formations with suitable geology and the demand for wastewater disposal associated with a mature and active oil and gas industry. In contrast, Class IID wells are a relatively small portion of Marcellus wastewater management in Pennsylvania (about 10% in 2013 and the first half of 2014) (PA DEP, 2015a) because the state has nine injection wells as of early 2015. Wastewater is generally transported out of state when being managed through injection into Class IID wells. The local availability of Class IID wells and the capacity to accept large volumes of wastewater may begin to be affected by recent state actions concerning seismic activity associated with injection (U.S. EPA, 2014f).

Table 8-3. Distribution of active Class IID wells across the United States.

Source: U.S. EPA (2015q).

State	Nearby basins with hydraulic fracturing	Number of active Class IID wells (2012-2014)	Average disposal rate per well (GPD/well)[a]	Total state disposal rate (MGD)

State	Nearby basins with hydraulic fracturing	Number of active Class IID wells (2012-2014)	Average disposal rate per well (GPD/well)[a]	Total state disposal rate (MGD)
AK	North Slope	45	182,000	8.2
OH	Appalachian	188	8,900	1.7
WV		66	7,180	0.47
PA		9	6,380	0.057
NY		10[b]	3,530	0.035
VA		12	17,500	0.21
TN		0	0	0
MD		0	0	0
NC	Multiple basins	0	0	0
KS	Cherokee, Anadarko, Arkoma	5,516	20,900	120
OK		4,622[c]	35,900	170
AR		611[d]	30,900	19
MO		11	1,270	0.014
CO	Denver-Julesburg, Green River, Piceance, Uinta	294	50,200	15
WY		330	-- [e]	-- [e]
UT		109	74,400	8.1
NE		113	18,100	2.0
TX	Fort Worth, Western Gulf, Permian, San Juan, Raton	7,876	54,200	430
NM		736	48,600	36
IN	Illinois	183	3,580	0.66
IL		1,054	-- [e]	-- [e]
KY		58	1,750	0.10
MI	Michigan	779[c]	16,600	13
CA	San Joaquin	826	77,800	64

State	Nearby basins with hydraulic fracturing	Number of active Class IID wells (2012-2014)	Average disposal rate per well (GPD/well)[a]	Total state disposal rate (MGD)
LA	TX-LA-MS Salt	2,448	42,100	100
MS		499	69,500	35
AL		85	44,200	3.8
ND	Williston	395	31,600	12
MT		199	31,100	6.2
SD		21	10,200	0.21
All other states (NV, FL, OR, IA, and WA)[f]		42	89,400	3.8
Total (not including missing states)		27,137	40,400	1,040

[a] Typical injection volumes per well are based on historical annual volumes for injection for disposal divided by the number of active Class IID wells during the same year (primarily data from 2007 to 2013).
[b] These wells are not currently permitted to accept unconventional oil and gas extraction wastewater.
[c] With the exception of Oklahoma and Michigan, wells on tribal lands have not been intentionally included. Wells on tribal lands may be counted if state databases contained them.
[d] Only 24 of the 611 active Class II wells in Arkansas are in the northern half of the state, close to the Fayetteville formation.
[e] Disposal rates and/or number of Class IID wells is unknown.
[f] These are states that have minimal oil and gas activity. The number of wells shown for these states may include all types of Class II wells (e.g., Class II enhanced recovery wells) and therefore is an upper estimate. All other states not listed in this table have minimal oil and gas activity and no active Class IID wells.

8.4.2. Centralized Waste Treatment Facilities

A CWT facility is generally defined as a facility that accepts industrial materials (hazardous, non-hazardous, solid, or liquid) generated at another facility (off-site) for treatment and/or recovery (EPA, 2000). (A POTW treats local municipal wastewater.) As a group, CWTs that accept oil and gas wastewater offer a wide variety of treatment capabilities and configurations. The fate of treated effluent at CWTs also varies, and can include the following: reuse in fracturing operations, direct discharge (to a receiving water under a National Pollution Discharge Elimination System (NPDES) permit), indirect discharge (to a POTW), or a combination of these. Zero discharge facilities do not discharge to either surface water or a POTW; effluent is generally used for reuse, although evaporation or land application may also be done. Some CWTs may be configured so that they only partially treat the waste stream if allowed by the end use (a reuse application that does not require TDS removal). Potential impacts on drinking water resources associated with treatment in CWTs will depend upon whether the CWT treats adequately for constituents of concern prior to discharge to surface water or to a POTW, and whether treatment residuals are managed appropriately.

Clean Water Act (CWA) regulations only apply to facilities that discharge treated wastewater to surface waters or POTWs. For zero-discharge facilities, Pennsylvania and Texas have adopted regulations to control permitting. PA DEP issues permits (General Permit WMGR123) that allow zero-discharge CWTs to treat and release water back to oil and gas industries for reuse (see the Eureka Resources Facility in Williamsport, PA listed in Table 8-7 as an example of a zero-discharge facility[1]). The Texas Railroad Commission (TXRRC) regulates and categorizes wastewater recycling facilities into different categories: off-lease commercial recycling facilities (capable of being moved from one location to another) and stationary commercial recycling facilities. The Texas regulations also promote oil and gas wastewater treatment for reuse and water sharing (see http://www.rrc.state.tx.us/rules/rule.php).

Wastewater from hydraulically fractured wells can be transported by truck or pipeline to and from a CWT (Easton, 2014); this may present a vulnerability for spills or leaks (see Chapter 7). The treated wastewater from CWTs may be integrated with other sources of water (for example, treated municipal wastewater, storm water drainage, or other treated industrial waste streams) for reuse applications (Easton, 2014).

8.4.2.1. Numbers and Locations of CWTs

Although there are CWTs serving hydraulic fracturing operations throughout the country, including the Barnett and Fayetteville shale plays plus oil fields in Texas and Wyoming, historically the majority have served Marcellus Shale operations. This is likely because the low availability of injection wells (Boschee, 2014) in Pennsylvania necessitates other forms of management. An EPA study (U.S. EPA, 2015q) identified 73 CWT facilities that have either accepted or plan to accept hydraulic fracturing wastewater (see Table 8-4). Of these, 39 are located in Pennsylvania. Most of these are zero-discharge facilities; they do not discharge to surface waters or POTWs, and they often do not include TDS removal. According to EPA research (U.S. EPA, 2015q), the number of CWT facilities serving operators in the Marcellus and Utica Shales has increased since the mid-2000s as the number of wells drilled in the Marcellus and Utica Shales has increased, growing from roughly five CWTs in 2004 to over 40 in 2013. A similar trend has been noted for the Fayetteville Shale region in Arkansas, where there are fewer Class IID wells available relative to the rest of the state (U.S. EPA, 2015q).

In other regions, a small number of newer facilities have emerged in the last several years, most often with TDS removal capabilities. In Texas, for example, two zero-discharge facilities are available to treat wastewater from the Eagle Ford (beginning in 2011 and 2013), both equipped with TDS removal, and one zero-discharge facility with TDS removal is located in the Barnett Shale region (operational beginning in 2008). In Wyoming, the four facilities in the region of the Mesaverde/Lance formations (operations beginning between 2006 and 2012; two zero-discharge and two with multiple discharge options) are all capable of TDS removal (U.S. EPA, 2015q).

[1] The facility is also permitted for indirect discharge to the Williamsport Sewer Authority.

Table 8-4. Number, by state, of CWT facilities that have accepted or plan to accept wastewater from hydraulic fracturing activities.

Source: U.S. EPA (2015q).

State	Formation(s) served where hydraulic fracturing occurs	Zero-discharge CWT facilities[a]		CWT facilities that discharge to surface water or POTW[a]		Discharging CWT facilities with multiple discharge options[a]		Total known facilities
		Non-TDS removal treatment	TDS removal treatment	Non-TDS removal treatment	TDS removal treatment	Non-TDS removal treatment	TDS removal treatment	
AR	Fayetteville	2	0	0	0	0	1	3
CO	Niobrara, Piceance Basin	3 (1)	0	0	0	0	0	3
ND	Bakken	0	1 (1)	0	0	0	0	1
OH	Utica, Marcellus	10 (7)	0	1	0	0	0	11
OK	Woodford	2	0	0	0	0	0	2
PA	Utica, Marcellus	23	7 (3)	6	0	0	3 (1)	39
TX	Eagle Ford, Barnett, Granite Wash	1	3	0	0	0	0	4
WV	Marcellus, Utica	4 (2)	0	0	0	1	1	6
WY	Mesaverde, Lance	0	2	0	0	0	2	4
Total		45	13	7	0	1	7	73

[a] Number of facilities also includes facilities that have not yet opened but are under construction, pending permit approval, or in the planning stages. Facilities that are not accepting process wastewater from hydraulic fracturing activities but plan to in the future are noted parenthetically.

Because few states maintain a comprehensive list of CWT facilities and the count provided by the EPA (U.S. EPA, 2015q) includes facilities that plan to accept unconventional oil and gas wastewaters, the data in Table 8-4 do not precisely reflect the number of facilities currently available for handling hydraulic fracturing wastewaters. Additional discussion of CWTs in unconventional oil and gas fields are reviewed in the literature for areas including the Barnett (Hayes and Severin, 2012b) and the Fayetteville (Veil, 2011) as well as other oil fields in Texas and Wyoming (Boschee, 2014, 2012). In addition, news releases and company announcements indicate that new wastewater treatment facilities are being planned (Greenhunter, 2014; Geiver, 2013; Purestream, 2013; Alanco, 2012; Sionix, 2011).

Based on oil and gas waste disposal information available from PA DEP (PA DEP, 2015a) dating back to 2009, the estimated volumes of Marcellus wastewater sent to CWTs range from approximately 113 Mgal (428 million L) in the latter half of 2009 and first half of 2010, to about 183 Mgal (693 million L) in 2011, and about 252 Mgal (954 million L) in 2013. These constitute about 52% of the total wastewater volume in 2009-2010, about 25% in 2011, and 20% in 2013, indicating that although total amounts of wastewater have increased (see Table 8-1), the percentage managed through CWTs has decreased.

Among the Marcellus wastewater sent to CWTs, an estimated 35% was sent to zero-discharge facilities in Pennsylvania (those with general permits) in the latter half of 2010, and 42% was sent to facilities with NPDES permits (indicating that they can discharge to surface waters). About 23% went to CWTs whose permit types were more difficult to ascertain, generally outside of Pennsylvania. By 2013, the portion sent to zero-discharge facilities had risen to 90%, with about 5% sent to CWTs with NPDES permits and 5% sent to CWTs whose discharge permit type is not clear. The high percentage sent to zero-discharge CWTs is consistent with the concerted focus on reuse in Pennsylvania, although CWTs with NPDES permits also often provide treated wastewater for reuse, further limiting discharges to surface waters. The waste records do not indicate if a CWT has more than one permit type.

8.4.2.2. Residuals Management

Certain treatment processes at CWTs produce liquid or solids residuals as a by-product of that process. The residuals produced depend on the constituents in the treated water and the treatment process used. Residuals can consist of sludges (from precipitation, filtration, settling units, and biological processes); spent media (media requiring replacement or regeneration from filtration, adsorption, or ion exchange processes); concentrated brines (from membrane processes and some evaporation processes); and regeneration and cleaning chemicals (from ion exchange, adsorption, and membrane processes) (Fakhru'l-Razi et al., 2009). Residuals from CWTs can constitute a considerable fraction of solid waste in an oil or gas production area. Chiado (2014) found that solid wastes from hydraulic fracturing in the Marcellus accounted for 5% of the weight of waste deposited in landfills in the area, with some area landfills reaching as high as 60% landfill mass coming from hydraulic fracturing activities.

Management of Solid Residuals

CWTs may apply additional treatment to solid residuals including thickening, stabilization (e.g., anaerobic digestion), and dewatering processes prior to disposal. The solid residuals are then typically sent to a landfill, land applied, or incinerated (Morillon et al., 2002). Pollutants may accumulate in sludge, which may limit land application as a disposal option. For example, wastes containing TENORMs can be problematic due to the possibility of radon emissions from the landfill (Walter et al., 2012). In some states, many landfills that are specifically permitted to accept TENORM have criteria written into their permits, including gamma exposure rate (radiation) levels and radioactivity concentration limits. Most non-hazardous landfills have limits on maximum radiation that can be accepted. For example, Pennsylvania requires alarms to be set at all municipal landfills, with a trigger set at 10 μR/hr above background radiation (Pa Code Title 25, Ch. 273.223

c). Texas sets a radioactivity limit, requiring that any waste disposed by burial contains less than 30 pCi/g radium or 150 pCi/g of other radionuclides (TX Code Ch 4 Section F Section 4.620). Some states have volumetric limitations on TENORM in their permits (e.g., Colorado).

Solid residual wastes have the potential to impact the quality of drinking water resources if contaminants leach to groundwater or surface water. In a recent study by PA DEP, radium was detected in leachate from 34 of 51 landfills, with radium-226 concentrations ranging from 54 to 416 pCi/L, and radium-228 ranging from 2.5 to 1,100 pCi/L (PA DEP, 2015b). Countess et al. (2014) studied the potential for barium, calcium, sodium, and strontium to leach from sludges generated at a CWT handling hydraulic fracturing wastewaters in Pennsylvania. Tests used various strong acid solutions (to simulate the worst case scenario) and weak acid digestions (to simulate environmental conditions). The extent of leaching varied by constituent and by fluid type; the data illustrate the possibility of leaching of these constituents from landfills.

Management of Liquid Residuals

Practices for management of liquid residual streams are generally the same as for untreated hydraulic fracturing wastewaters, although the reduced volumes tend to lower costs (Hammer and VanBriesen, 2012). Concentrations of contaminants, however, will be higher. Liquids mixed with other wastes can be disposed of in landfills if the liquid concentration is low enough. If the liquid is not injected into a disposal well, treatment to remove salts would be required for surface water discharge to meet NPDES permit requirements and protect the water quality for downstream users (e.g., drinking water utilities) (see Section 8.6). Because some constituents of concentrated residuals can pass through or impact municipal wastewater treatment processes (Linarić et al., 2013; Hammer and VanBriesen, 2012), these residuals may not be appropriate for discharge to a POTW. Elevated salt concentrations, in particular, can reduce or inhibit microbiological treatment at municipal wastewater systems such as activated sludge treatment (Linarić et al., 2013).

8.4.3. Water Reuse for Hydraulic Fracturing

Water reuse in hydraulic fracturing operations has increased in recent years, with wastewaters being used to formulate hydraulic fracturing fluids for subsequent fracturing jobs (Boschee, 2014, 2012; Gregory et al., 2011; Rassenfoss, 2011). Wastewater may be reused after some form of treatment (sometimes only settling), depending on the reuse water quality requirements, and it may be supplied for use in hydraulic fracturing through various routes. Reused water is discussed in Chapter 4 of this report (Water Acquisition) as well as in this chapter, though in a different context. The water reuse rate described in this chapter is the amount or percentage of generated wastewater that is managed by being provided to operators for use in additional hydraulic fracturing operations. In contrast, Chapter 4 discusses reused wastewater as a source water and as one part of the base fluid for new fracturing fluid.

Hydraulic fracturing wastewater reuse reduces costs associated with other forms of wastewater management, and the economic benefits and feasibility of reuse can be expected to figure into ongoing wastewater management decisions. However, although reuse minimizes other forms of wastewater management on a local and short-term basis (e.g., those involving direct or indirect discharge to surface waters), reuse can result in the accumulation of dissolved solids (e.g., salts and TENORMs) as the process returns water to the subsurface. For example, data from a PA DEP study (PA DEP, 2015b) suggests that hydraulic fracturing fluids that include reused wastewater already contain radium-226 and radium-228. Eventually, wastewaters with a component that has been reused more than once will need to be definitively managed, either through treatment or injection. Residuals from treatment will also require proper management to avoid potential impacts on water resources (see Section 8.4.2.2) (Kappel et al., 2013).

8.4.3.1. Factors in Considering Reuse

In making the decision whether to manage wastewater via reuse, operators have several factors to consider (Slutz et al., 2012; NPC, 2011a):

- Wastewater generation rates compared to water demand for future fracturing operations,
- Wastewater quality and treatment requirements for use in future operations,
- The costs and benefits of wastewater management for reuse compared with other management strategies,
- Available infrastructure and treatment technologies, and
- Regulatory considerations.

Among these factors, costs may be the most significant driver, weighing the costs of transportation from the generating well to the treatment facility and to the new well against the costs for transport to alternative locations (a disposal well or CWT). Trucking large quantities of water can be relatively expensive (from $0.50 to $8.00 per barrel), rendering on-site treatment technologies and reuse potentially economically competitive in some settings (Dahm and Chapman, 2014; Guerra et al., 2011). Also, logistics, including proximity of the water sources for aggregation, may be a factor in implementing reuse. For example, Boschee (2014) notes that in the Permian Basin, older conventional wells are linked by pipelines to a central disposal facility, facilitating movement of treated water to areas where it is needed for reuse.

Regulatory factors may facilitate reuse. In 2013, the Texas Railroad Commission adopted rules intended to encourage statewide water conservation. These rules facilitate reuse by eliminating the need for a permit when operators reuse on their own lease or transfer the fluids to another operator for use in hydraulic fracturing (Rushton and Castaneda, 2014). Data for the years after 2013 will allow evaluation of whether reuse increases.

Recommended compositional ranges for base fluid may shift in the future as fracturing fluid technology continues to develop. Development of fracturing mixture additives that are brine-tolerant have allowed for the use of high TDS wastewaters (up to tens of thousands of mg/L) for reuse in fracturing (Tiemann et al., 2014; GTI, 2012; Minnich, 2011). Some new fracturing fluid systems are claimed to be able to tolerate salt concentrations exceeding 300,000 mg/L (Boschee, 2014). This greater flexibility in acceptable water chemistry can facilitate reuse both logistically and economically by reducing treatment needs. Additional discussion of the water quality feasible for reuse and examples of recommended constituent concentrations are included in Appendix F.

Reuse rates may also fluctuate with changes in the supply and demand of treated wastewater and the availability of fresh water. Flowback may be preferable to later-stage produced water for reuse because it is typically generated in larger quantities from a single location as opposed to water produced later on, which is generated in smaller volumes over time from many different locations. Flowback water also tends to have lower TDS concentrations than later-stage produced water; in the Marcellus, TDS has been shown to increase from tens of thousands to about 100,000 mg/L during the first 30 days (Barbot et al., 2013; Maloney and Yoxtheimer, 2012) (see Chapter 7). The changing production rate and quality of wastewaters generated in a region as more wells go into production need to be taken into account, as well as possible decreases in the demand for reused water as plays mature (Lutz et al., 2013; Hayes and Severin, 2012b; Slutz et al., 2012).

8.4.3.2. Reuse Rates

Reliable information on reuse practices throughout the United States is hampered by a limited amount of data that are available and represent different regions of the country. In Table 8-5, estimates have been compiled from various literature sources. Reuse rates are highest in the Appalachian Basin, associated primarily with the Marcellus Shale. Documentation of reuse practices is also more readily available for that region than for other parts of the country.

A number of studies have estimated reuse rates for Marcellus wastewater. Although the reported values can differ substantially (see Table 8-5), the data point to a steep increase in reuse since 2008, with rates increasing from 0% to 10% in 2008 to upwards of 90% in 2013. As an example, an analysis of waste disposal information from the PA DEP for Marcellus wells in Pennsylvania (Hansen et al., 2013) reports an increase in reuse from 9% (7.17 million gal or 27.1 million L) of total wastewater volumes in 2008 to 56% (343.79 million gal or 1.3014 billion L) in 2011. During that same timeframe, the authors report that disposal via brine/industrial waste treatment plants increased from 32% in 2008 to 70% in 2009, and then declined to 30% in 2011. Because some industrial waste treatment plants can treat wastewater for reuse, some of the volumes indicated by Hansen et al. (2013) as managed by this route may have ultimately been used for fracturing, meaning that the 56% value for 2011 is most likely an underestimate.

Table 8-5. Estimated percentages of reuse of hydraulic fracturing wastewater.

Play or Basin	Source and Year	2008	2009	2010	2011	2012	2013
East Coast							
Marcellus, PA	Rahm et al. (2013)	9	8	25 – 48	67 - 80		
Marcellus, PA	Ma et al. (2014)		15 - 20				90
Marcellus, PA	Shaffer et al. (2013)					90	
Marcellus, WV	Hansen et al. (2013)			88	73	65 (partial year)	
Marcellus, PA	Hansen et al. (2013)	9	6	20	56		
Marcellus, PA	Maloney and Yoxtheimer (2012)				71.6		
Marcellus, PA	Tiemann et al. (2014)				72	87	

Play or Basin	Source and Year	2008	2009	2010	2011	2012	2013
Marcellus, PA	Rassenfoss (2011)			~67 overall (general estimate) 96 (one specific company)			
Marcellus, PA	Wendel (2011)			75-85	90		
Marcellus, PA	Lutz et al. (2013)	13 (prior to 2011)			56		
Marcellus, PA (SW region)	Rahm et al. (2013)	~10	~15	~25-45	~70-80		
Marcellus, PA (NE region)	Rahm et al. (2013)	0	0	~55-70	~90-100		
Marcellus, PA	Rahm and Riha (2014)				55-80 (general estimate – appears to cover recent years)		
Gulf Coast & Midcontinent							
Fayetteville	Veil (2011)			20 (single company target)			
West Permian	Nicot et al. (2012)				0		
Midland Permian	Nicot et al. (2012)				2		
Anadarko	Nicot et al. (2012)				20		
Barnett	Nicot et al. (2012)				5		
Barnett	Rahm and Riha (2014)				5 (general estimate – appears to cover recent years)		
Eagle Ford	Nicot and Scanlon (2012)				0	20 (estimate based on interviews)	
East Texas	Nicot and Scanlon (2012)				5		
Haynesville	Argonne National Laboratory (2014)						0
Haynesville	Rahm and Riha (2014)				5 (general estimate – appears to cover recent years)		
West Coast & Upper Plains							
Bakken	Argonne National Laboratory (2014)						0

According to Maloney and Yoxtheimer (2012), about 331 million gal (7.9 million bbl or 1.25 billion L) of flowback and about 381 million gal (about 9.1 million bbl or 1.4 billion L) of produced water (excluding flowback) were generated in the Marcellus in 2011. For flowback and produced water combined, about 72% was reused. Of the flowback, 90% was managed through reuse (other than road spreading). Of produced brine water, 55.7% was reused (with 11.6% treated in CWTs and 27.8% injected into Class IID wells in Ohio). Reuse is higher in the northeastern part of the Marcellus; in the southwestern portion, easier access to Class IID wells in Ohio makes disposal by injection more feasible (Rahm et al., 2013).

Data from Marcellus wastewater management reports submitted to PA DEP (PA DEP, 2015a) were compiled for this assessment; the data suggest that rates of reuse for hydraulic fracturing (as indicated by a waste disposal method of either "Reuse Other than Road Spreading" or a zero-discharge CWT) increased from about 28% in the second half of 2010 to about 60% in 2011, 83% in 2013, and 89% in the first half of 2014. These values may be underestimates because wastewater treated at facilities with NPDES permits can be provided to operators for reuse, and the permit types for some facilities could not be determined. Among the forms of reuse, on-site reuse ("Reuse Other than Road Spreading") has risen steadily over the past few years, from about 8% in the second half of 2010 to about 48% in 2011, 62% in 2012, and nearly 70% in the first half of 2014.

Outside of the Marcellus region, a lower percentage of wastewater from hydraulic fracturing operations is reused. According to published literature, in Texas in 2011, 0% to 5% of wastewater was reused in most basins, with the exception of the Anadarko Basin (20%) (Nicot and Scanlon, 2012); see Table 8-5. Ma et al. (2014) note that only a small amount of reuse is occurring in the Barnett Shale. Reuse has not yet been pursued aggressively in New Mexico or in the Bakken (North Dakota) (Argonne National Laboratory, 2014; LeBas et al., 2013). Other sources, however, indicate growing interest in reuse, as evidenced in specialized conferences (e.g., "Produced Water Reuse Initiative 2014" on produced water reuse in Rocky Mountain oil and shale gas plays), and available state-developed information on reuse (e.g., fact sheet by the Colorado Oil and Gas Conservation Commission) (Colorado Division of Water Resources; Colorado Water Conservation Board; Colorado Oil and Gas Conservation Commission, 2014). The fact sheet discusses piping and trucking wastewater to CWTs in the Piceance Basin to treat for reuse.

8.4.4. Evaporation

In drier climates of the western United States, natural evaporation may be an option for treatment of hydraulic fracturing wastewater (see Figure 8-6). Production data from the California Department of Conservation's Division of Oil, Gas, and Geothermal Resources (DOGGR) (California Department of Conservation, 2015), for example, lists "evaporation-percolation" as the management method for 23% to 30% of the wastewater from Kern County over the last few years. However, data on volumes of wastewater managed are not readily available for all states where this practice is employed.

Evaporation is a simple water management strategy that consists of transporting wastewater to a pond or pit with a large surface area and allowing passive evaporation of the water from the surface (Clark and Veil, 2009). The rate of evaporation depends on the quality of the wastewater as well as the size, depth, and location of the pond. Evaporation also depends on local humidity, temperature, and wind (NETL, 2014). The residual brine or solid can be disposed of in an underground injection well or landfill (see Section 8.4.3.2 for more details). In colder, dry climates, a freeze-thaw evaporation method has been used to purify water from oil and gas wastewater (Boysen et al., 1999).

Figure 8-6. Lined evaporation pit in the Battle Creek Field (Montana).

Source: DOE (2006). Permission from ALL Consulting.

Alternatively, operators may transport wastewater by truck to an off-site commercial facility. Commercial evaporation facilities exist in Colorado, Utah, New Mexico, and Wyoming (NETL, 2014; DOE, 2004). Nowak and Bradish (2010) described the design, construction, and operation of two large commercial evaporation facilities in Southern Cross, Wyoming and Danish Flats, Utah. Each facility includes 14,000-gal (53,000 L) three-stage concrete receiving tanks, a sludge pond, and a series of five-acre (20,234 m^2) evaporation ponds connected by gravity or force-main underground piping. The Wyoming facility, which opened in 2008, consists of two ponds with a total capacity of approximately 84 million gal (2 million bbl or 320 million L). The Utah facility, open since 2009, consists of 13 ponds with a total capacity of 218.4 million gal (5.2 million bbl or 826.7 million L). Each facility receives 420,000 to 1.47 million gal (10,000 to 35,000 bbl or 1.6 million to 5.56 million L) per day of wastewater from oil and gas production companies in the area. Evaporation pits are subject to state regulatory agency approval and must meet state standards for water quality and quantity (Boysen et al., 2002). Impacts on drinking water resources from evaporation pits might arise if a pit is breached due to extreme weather or other factors affecting infrastructure and if leaking wastewater reaches a surface water body; such events as related specifically to evaporation pits appear not to have not been evaluated in the literature, and their prevalence is unknown.

8.4.5. Publicly Owned Treatment Works

Prior to the development of unconventional resources, POTWs were used to treat wastewater and other wastes from conventional oil and gas operations in some eastern states. Although this is not a common treatment method for oil and gas wastes in the United States, the small number of injection wells for waste disposal in Pennsylvania drove the need for disposal alternatives (Wilson and Vanbriesen, 2012). When development of the Marcellus Shale began, POTWs continued to be used to treat wastewater, including wastes originating from new unconventional oil and gas wells (Kappel et al., 2013; Soeder and Kappel, 2009). However, unconventional wastewater from the Marcellus region is difficult to treat at POTWs due to elevated concentrations of halides, heavy metals, organic compounds, radionuclides, and salts (Lutz et al., 2013; Schmidt, 2013). Most of these constituents have the potential to pass through the unit treatment processes commonly used in POTWs and can be discharged into receiving waters (Cusick, 2013; Kappel et al., 2013). In addition, research has found that sudden, extreme salt fluctuations can disturb POTW biological treatment processes (Linarić et al., 2013; Lefebvre and Moletta, 2006). In order to meet NPDES requirements, POTWs used to blend the hydraulic fracturing wastewater with incoming municipal wastewater. For example, Ferrar et al. (2013) note that, per PA DEP requirements, one facility could only accept up to 1% of their influent volume from unconventional oil and gas wastewater per day.

The annual reported volume of oil and gas produced wastewater treated at POTWs in the Marcellus Shale region peaked in 2008 and has since declined to virtually zero (see Figure 8-7). This decline has been attributed to stricter discharge limits for TDS for POTWs in Pennsylvania and widespread voluntary compliance on behalf of oil and gas operators with the May 2011 request from PA DEP to cease sending Marcellus Shale wastewater to 15 treatment plants (including both POTWs and CWTs) by May 19, 2011 (Rahm et al., 2013). To comply with the request, the oil and gas industry in Pennsylvania accelerated the transition of wastewater deliveries from POTWs to CWTs for better removal of metals and suspended solids (Schmidt, 2013). However, treated effluent from CWTs may be delivered to POTWs for additional treatment assuming treatment processes at POTWs are not adversely affected and the POTWs can continue to meet NPDES discharge limits (Hammer and VanBriesen, 2012). General Pretreatment Regulations and State and local regulations typically govern the pre-treated water volumes and qualities that can be accepted by the POTW.

Although operators stopped sending Marcellus Shale wastewater to POTWs in May of 2011, conventionally produced wastes have continued to be processed at POTWs in Pennsylvania, although at small volumes (29 Mgal and 20 Mgal for the years 2010 and 2011, respectively) (Wilson and Vanbriesen, 2012).

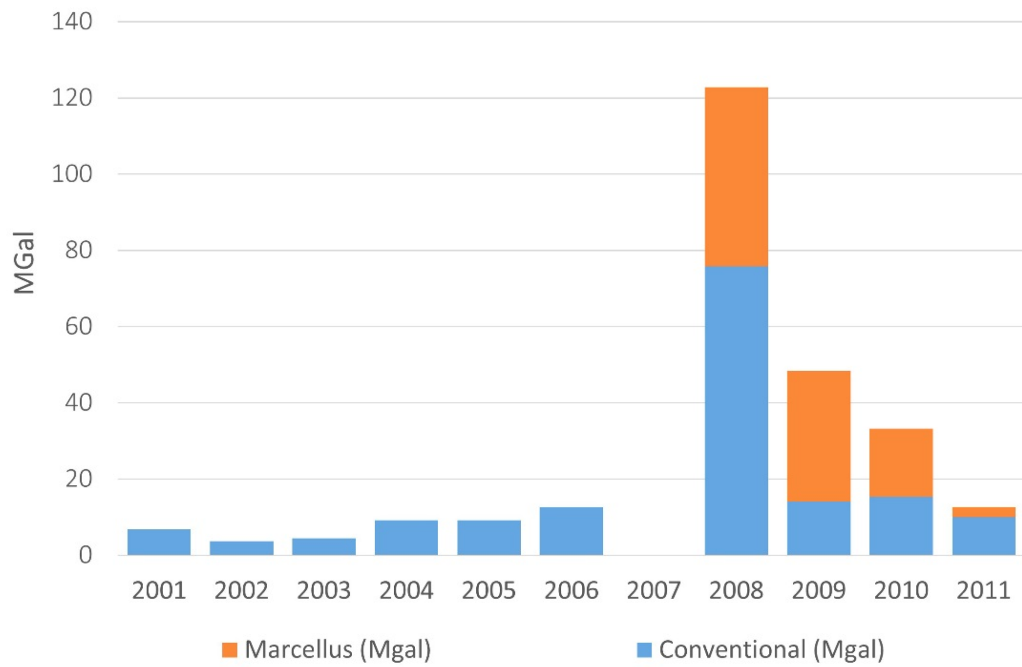

Figure 8-7. Oil and gas wastewater volumes discharged to POTWs from 2001-2011 in the Marcellus Shale.

Source: Lutz et al. (2013).

At least one study has evaluated POTW effluent chemistry before and after the cessation of treatment of hydraulic fracturing wastewater. Ferrar et al. (2013) collected effluent samples from two POTWs and one CWT facility in Pennsylvania before and after the 2011 PA DEP request. Results from POTW effluent samples collected while the facilities were still treating Marcellus Shale wastewater showed that concentrations of several analytes (barium, manganese, strontium, TDS, and chloride) were greater than various drinking water and surface water criteria (i.e., EPA MCLs and secondary MCLs for drinking water, surface water quality standards for aquatic life, and/or surface water standards for human consumption of aquatic organisms). Results for effluent samples collected after the POTWs stopped receiving Marcellus wastewater showed a statistically significant decrease in the concentrations of several of these constituents. In particular, one of the two POTWs showed a decrease in average barium concentration from 5.99 mg/L to 0.141 mg/L, a decrease in the average strontium concentration from 48.3 mg/L to 0.236 mg/L, and a decrease in the average bromide concentration from 20.9 mg/L to <0.016 mg/L. Influent concentrations at the other POTW were lower (0.55 mg/L for barium, 1.63 mg/L for strontium, and 0.60 for bromide), but significant decreases in these constituents were also seen in the effluents (0.036 mg/L barium, 0.228 mg/L strontium, and 0.119 for bromide); this POTW had continued to accept conventional oil and gas wastewater. The authors conclude that the decreases in the concentrations of the various constituents indicate that the elevated concentrations in the first samplings can be attributed to the contribution of wastewater from unconventional natural gas development.

8.4.6. Other Management Practices and Issues

Additional strategies for wastewater management in some states include discharging to surface waters and land application. Wastewater from CBM fracturing and production, in particular, generally has lower TDS concentrations than wastewater from other types of unconventional plays and lends itself more readily to beneficial use. Below is a discussion of these other management practices.

8.4.6.1. Land Application, Including Road Spreading

Land application has been done using brines from conventional oil and gas production. Road spreading can be done for dust control or de-icing. Although recent data are not available, an American Petroleum Institute (API) survey estimated that approximately 75.6 million gal (1.8 million bbl or 286 million L) of wastewater was used for road spreading in 1995 (API, 2000). The API estimate does not specifically identify hydraulic fracturing wastewater. There is no current nationwide estimate of the extent of road spreading using hydraulic fracturing wastewater.

Road spreading with hydraulic fracturing wastewater is regulated primarily at the state level (Hammer and VanBriesen, 2012) and is prohibited in some states. For example, with annual approval of a plan to minimize the potential for pollution, PA DEP allows spreading of brines from conventional wells for dust control or road stabilization. Hydraulic fracturing flowback, however, cannot be used for dust control and road stabilization (PA DEP, 2011b). In West Virginia, use of gas well brines for roadway de-icing is allowed per a 2011 memorandum of agreement between the West Virginia Division of Highways and the West Virginia Department of Environmental Protection, but the use of "hydraulic fracturing return fluids" is not permitted (Tiemann et al., 2014; West Virginia DEP, 2011).

Concerns about road application center on contaminants such as barium, strontium, and radium. A report from PA DEP analyzed several commercial rock salt samples and compared results with contaminants found in Marcellus Shale flowback samples; the results noted elevated barium, strontium, and radionuclide levels in Marcellus Shale brines compared with commercial rock salt (Titler and Curry, 2011). Another study found increases in metals (radium, strontium, calcium, and sodium) in soils ranging from 1.2 to 6.2 times the original concentration (for radium and sodium, respectively), attributed to road spreading of wastewater from conventional oil and gas wells for de-icing purposes (Skalak et al., 2014).

Potential impacts on drinking water resources from road spreading, have been noted by Tiemann et al. (2014) and Hammer and VanBriesen (2012). These include potential effects of runoff on surface water, or migration of brines to groundwater. Snowmelt may carry salts or other chemicals from the application site, with the possibility of transport increasing if application rates are high or rain occurs soon after application (Hammer and VanBriesen, 2012). Research on the impacts of conventional road salt application has documented long-term salinization of both surface water and ground water in the northern United States; by the 1990s, 24% of public supply wells in the Chicago area had chloride concentrations exceeding 100 mg/L (Kelly, 2008; Kaushal et al., 2005). When conventional oil field brine was used in a controlled road spreading experiment, elevated chloride concentrations were detected in shallow ground water (531 ppm in winter and 1,360 ppm in summer (Bair and Digel, 1990). The amount of salt contributed to drinking water resources due to road application of hydraulic fracturing wastewaters has not been quantified.

In managing solid wastes from oil and gas production, a study on land application of oilfield scales and sludges suggested that radium in samples became more mobile after incubation with soil under moist conditions, due to microbial processes and interactions with the soil and water (Matthews et al., 2006). Overall, potential effects from land application on drinking water resources are not well understood.

Additionally, drill cuttings must be managed; in some places they are left in the reserve pit (pit for waste storage), allowed to dry and, buried on-site (Kappel et al., 2013). More, commonly, however, drill cuttings are disposed of in landfills (Chiado, 2014); about half of Marcellus drill cuttings are disposed of in Pennsylvania, while the rest are trucked to Ohio or West Virginia (Maloney and Yoxtheimer, 2012).

8.4.6.2. Management of Coalbed Methane Wastewater

Wastewater from CBM wells can be managed like other hydraulic fracturing wastewater discussed above. However, the wastewater from CBM wells can also be of higher average quality (e.g., lower TDS content) than wastewater from other hydraulically fractured wells, which makes it more suitable for certain management practices and uses. A number of management strategies have been proposed or implemented, with varying degrees of treatment required depending on the quality of the wastewater and the requirements of the intended use (Hulme, 2005; DOE, 2003, 2002). Although specific volumes managed through the various practices below are not well documented, qualitative information and considerations for feasibility are available and presented below.

CBM wastewater quality, which can range from an average of nearly 1,000 mg/L TDS in the Powder River Basin to an average of about 4,700 mg/L in the San Juan Basin (see Appendix Table E-3), plays a large role in how the wastewater is managed. In basins with higher TDS such as the San Juan, Uinta, and Piceance, nearly all the wastewater is disposed of in injection wells. Wastewater may also be injected for aquifer storage and recovery, with the intention of later recovering the water for some other use (DOE, 2003).

Discharge to rivers and streams for wildlife, livestock, and agricultural use, a management option governed by the CWA, may be permitted in some cases. To be discharged, the wastewater must meet technology-based limitations established by the permit authority and any applicable water quality water quality standards. Direct discharge to streams (with or without treatment) is possible where wastewater is of higher quality. This is a more common method of wastewater management in basins such as the Raton Basin in Colorado and the Tongue River drainage of the Powder River Basin in Montana (NRC, 2010).

Agricultural uses include livestock watering, crop irrigation, and commercial fisheries. Livestock watering with CBM wastewater is a common practice, and irrigation is an area of active research (e.g., Engle et al., 2011; NRC, 2010). Irrigation with treated CBM wastewater would be most suitable on coarse-textured soils, for cultivation of salt-tolerant crops (DOE, 2003). NRC (2010) remarks that "use of CBM produced water for irrigation appears practical and sustainable," provided that appropriate measures are taken such as selective application, dilution or blending, appropriate timing, and rehabilitation of soils. Approximately 13% of CBM wastewater in the Powder River Basin in Wyoming, and 26% to 30% in Montana, is used for irrigation (NRC, 2010).

As noted above, a degree of treatment is needed for some uses. Plumlee et al. (2014) examined the feasibility, treatment requirements, and cost of several hypothetical uses for CBM wastewater. In several cases, costs for these uses were projected to be comparable to or less than estimated disposal costs. In one case study CBM wastewater for stream augmentation or crop irrigation was estimated to cost between $0.26 and $0.27 a barrel and disposal costs ranged from $0.01 per barrel (pipeline collection system with impoundment) to $2.00 per barrel (hauling for disposal or treatment).

The applicability of particular uses may be limited by ecological and regulatory considerations, as well as the irregular nature of CBM wastewater production (voluminous at first, and then declining and halting after a period of years). Legal issues, including overlapping jurisdictions at the state level and, in western states, senior water rights claims in over-appropriated basins, may also determine use of CBM wastewater (Wolfe and Graham, 2002).

8.4.6.3. Other Documented Uses of Hydraulic Fracturing Wastewater

Uses of wastewater from shales or other hydraulically fractured formations face many of the same possibilities and limitations as those associated with wastewater from CBM operations. The biggest difference is in the quality of the water. Wastewaters vary widely in water quality, with TDS values from shale sand tight sand formations ranging from less than 1,000 mg/L TDS to hundreds of thousands of mg/L TDS (DOE, 2006). Wastewaters on the lower end of the TDS spectrum could be reused in many of the same ways as CBM wastewaters, depending on the concentrations of potentially harmful constituents and applicable federal, state, and local regulations. High TDS wastewaters have more limited uses, and pre-treatment may be necessary (Shaffer et al., 2013; Guerra et al., 2011; DOE, 2006).

Documented potential uses for wastewater in the western United States include livestock watering, irrigation, supplementing stream flow, fire protection, road spreading, and industrial uses, with each having their own water quality requirements and applicability (Guerra et al., 2011). Guerra et al. summarized the least conservative TDS standards for five possible uses in the western United States that include 500 mg/L for drinking water (the secondary MCL), 625 mg/L for groundwater recharge, 1,000 mg/L for surface water discharge, 1,920 mg/L for irrigation, and 10,000 mg/L for livestock watering. The authors estimated that wastewater from 88% of unconventional wells in the western United States could be used for livestock watering without treatment for TDS removal based on a maximum TDS concentration of 10,000 mg/L. Wastewater from 10% of unconventional wells were estimated to meet the criterion of 1,000 mg/L TDS for surface water discharge (Guerra et al., 2011).

A 2006 Department of Energy (DOE) study points out that the quality necessary for use in agriculture depends on the plant or animal species involved. Other important factors include the sodium adsorption ratio and concentrations of TDS, calcium, magnesium, and other constituents

(DOE, 2006). The authors note that in the Bighorn Basin in Wyoming, low salinity wastewater is used for agriculture and livestock watering after minimal treatment to remove oil and grease (DOE, 2006).

8.5. Summary and Analysis of Wastewater Treatment

A variety of individual treatment techniques and combinations of techniques may be employed for removal of hydraulic fracturing wastewater constituents of concern. These include methods commonly employed in municipal wastewater treatment as well as more advanced processes such as desalination. Treatment technologies are selected based on the water quality of the wastewater to be treated and the effluent concentration required for the intended management method(s) (i.e., reuse, discharge to POTW, and discharge to surface water body). For example, if reuse is planned, the level of treatment will depend on the water quality needed to formulate the new fracturing fluid.

This section discusses treatment technologies that are most effective for removing specific hydraulic fracturing wastewater constituents. It provides information on the unit processes appropriate for treating different types of constituents as well as challenges associated with their use. Considerations when designing a treatment system are also discussed for both centralized and on-site (i.e., mobile) facilities.

8.5.1. Overview of Treatment Processes for Hydraulic Fracturing Wastewater

This section provides a brief overview of the treatment technologies used to treat hydraulic fracturing wastewater; Appendix F provides more in-depth descriptions of these technologies.

The most basic treatment need for oil and gas wastewaters, including those from hydraulic fracturing operations is separation to remove suspended solids and oil and grease, done using basic separation technologies (e.g., hydrocyclones, dissolved air or induced gas flotation, media filtration, or biological aerated filters). Other treatment processes that may be used include media filtration after chemical precipitation for hardness and metals (Boschee, 2014), adsorption technologies, including ion exchange (organics, heavy metals, and some anions) (Igunnu and Chen, 2014), a variety of membrane processes (microfiltration, ultrafiltration, nanofiltration, RO), and distillation technologies. In particular, advanced processes such as RO or distillation methods (e.g., mechanical vapor recompression (MVR)) are needed for significant reduction in TDS (Drewes et al., 2009; LEau LLC, 2008; Hamieh and Beckman, 2006). An emerging technology is electrocoagulation, which has been used in mobile treatment systems to treat hydraulic fracturing wastewaters (Halliburton, 2014; Igunnu and Chen, 2014). Removal efficiencies for hydraulic fracturing wastewater constituents by treatment technology are provided in Appendix F.

8.5.2. Treatment of Hydraulic Fracturing Waste Constituents of Concern

The constituents prevalent in hydraulic fracturing wastewater include suspended solids, TDS, anions (e.g., chloride, bromide, and sulfate), metals, radionuclides, and organic compounds (see Section 8.3 and Chapter 7). If the end use of the wastewater necessitates treatment, a variety of technologies can be employed. This section discusses effective unit processes for removing these constituents and provides examples of treatment processes being used in the field as well as emerging technologies. Table 8-6 provides an overview of influent and effluent results at various CWTs for the constituents of concern listed in this section and the specific technology(ies) used to remove them.

Table 8-6. Studies of removal efficiencies and influent/effluent data for various processes and facilities.

Constituents of concern	Location and results				
	Pinedale Anticline Water Reclamation Facility, Wyoming (Shafer, 2011)	**Maggie Spain Water-Recycling Facility, Barnett Shale, Texas (Hayes et al., 2014)**	**Judsonia, Sunnydale, Arkansas (U.S. EPA, 2015f)**	**9-month study treating Marcellus Shale waste using thermal distillation (Boschee, 2014; Bruff and Jikich, 2011)**	**San Ardo Water Reclamation Facility, San Ardo, California (Conventional oil and gas) (Dahm and Chapman, 2014; Webb et al., 2009)**
TSS	Results not reported.	90% Inf. = 1,272 mg/L Eff. = 9 mg/L Chemical oxidation, coagulation, and clarification	No influent data. Eff.: <4 mg/L Meets NPDES Permit Settling, biological treatment, and induced gas flotation	>90% Inf.: 35 to 114 mg/L Eff.: <3 to 3 mg/L 100 micron mesh bag filter	Results not reported.
TDS	>99% Inf. = 8,000 to 15,000 mg/L Eff. = 41 mg/L RO	99.7% Inf. = 49,550 mg/L Eff. = 171 mg/L MVR (3 units in parallel)	Results not reported. MVR	98% Inf.: 22,350 to 37,600 mg/L Eff.: 9 to 400 mg/L Thermal distillation	97% Inf. = 7,000 mg/L Eff. = 180 mg/L Ion exchange softening and double-pass RO

Constituents of concern	Location and results				
	Pinedale Anticline Water Reclamation Facility, Wyoming (Shafer, 2011)	**Maggie Spain Water-Recycling Facility, Barnett Shale, Texas (Hayes et al., 2014)**	**Judsonia, Sunnydale, Arkansas (U.S. EPA, 2015f)**	**9-month study treating Marcellus Shale waste using thermal distillation (Boschee, 2014; Bruff and Jikich, 2011)**	**San Ardo Water Reclamation Facility, San Ardo, California (Conventional oil and gas) (Dahm and Chapman, 2014; Webb et al., 2009)**
Anions	Chloride: >99% Inf. = 3,600 to 6,750 mg/L Eff. = 18 mg/L Sulfate: 99% Inf. = 10 to 100 mg/L Eff. = non-detect RO	Sulfate: 98% Inf. = 309 mg/L Eff. = 6 mg/L Chemical oxidation, coagulation, clarification, and MVR	Sulfate: No influent data. Eff.: 12 mg/L Meets NPDES Permit MVR	Bromide: >99% Inf.: 101 to 162.5 mg/L Eff.: <0.1 to 1.6 mg/L Chloride: 98% Inf.: 9,760 to 16,240 mg/L Eff.: 2.9 to 184.2 mg/L Sulfate: 93% Inf.: 20.4 to <100 mg/L Eff.: <1 to 2.2 mg/L	Chloride: >99% Inf. = 3,400 mg/L Eff. = 11 mg/L Double-pass RO Sulfate: 6% Inf. = 133 mg/L Eff. = 125 mg/L
				Fluoride: 96% Inf.: <2 to <20 mg/L Eff.: <0.2 to 0.42 mg/L Thermal distillation	Sulfuric acid is added after RO to neutralize the pH so no sulfate removal is expected.

Constituents of concern	Location and results				
	Pinedale Anticline Water Reclamation Facility, Wyoming (Shafer, 2011)	**Maggie Spain Water-Recycling Facility, Barnett Shale, Texas (Hayes et al., 2014)**	**Judsonia, Sunnydale, Arkansas (U.S. EPA, 2015f)**	**9-month study treating Marcellus Shale waste using thermal distillation (Boschee, 2014; Bruff and Jikich, 2011)**	**San Ardo Water Reclamation Facility, San Ardo, California (Conventional oil and gas) (Dahm and Chapman, 2014; Webb et al., 2009)**
Metals	Boron: 99% Inf. = 15 to 30 mg/L Eff. = non-detect Ion exchange	Iron: >99% Inf. = 28 mg/L Eff. = 0.1 mg/L For iron, 90% attributed to chemical oxidation, coagulation, and clarification Boron: 98% Inf. = 17 mg/L Eff. = 0.4 mg/L Barium: >99% Inf. = 15 mg/L Eff. = 0.1 mg/L Calcium: >99% Inf. = 2,916 mg/L Eff. = 3.2 mg/L	Cobalt: No influent data. Eff.: <0.007 mg/L Tin: No influent data. Eff.: <0.1 mg/L Arsenic: No influent data. Eff.: <0.001 mg/L Cadmium: No influent data. Eff.: <0.0001 mg/L Chromium: No influent data. Eff.: <0.007 mg/L Copper: No influent data. Eff.: <0029 mg/L	Copper: >99% Inf. = <0.2 to <1.0 mg/L Eff. = <0.02 to <0.08 mg/L Zinc: inf below detect Inf. = <0.2 to <1.0 mg/L Eff. = <0.02 to 0.05 mg/L Barium: >99% Inf. = 260.5 to 405.5 mg/L Eff. = <0.1 to 4.54 mg/L Strontium: 98% Inf. = 233 to 379 mg/L Eff. = 0.026 to 3.93 mg/L Iron: Inf. = 13.9 to 22.9 mg/L Eff. = <0.02 to 0.06 mg/L Manganese: 98% Inf. = 2 to 2.9 mg/L Eff. = <0.02 to 0.04 mg/L	Sodium: 98% Inf. = 2,300 mg/L Eff. = 50 mg/L Boron: >99% Inf. = 26 mg/L Eff. = 0.1 mg/L RO with elevated influent pH

Constituents of concern	Location and results				
	Pinedale Anticline Water Reclamation Facility, Wyoming (Shafer, 2011)	**Maggie Spain Water-Recycling Facility, Barnett Shale, Texas (Hayes et al., 2014)**	**Judsonia, Sunnydale, Arkansas (U.S. EPA, 2015f)**	**9-month study treating Marcellus Shale waste using thermal distillation (Boschee, 2014; Bruff and Jikich, 2011)**	**San Ardo Water Reclamation Facility, San Ardo, California (Conventional oil and gas) (Dahm and Chapman, 2014; Webb et al., 2009)**
Metals, cont.		Magnesium: >99% Inf. = 316 mg/L Eff. = 0.4 mg/L Sodium: >99% Inf. = 10,741 mg/L Eff. = 14.3 mg/L Strontium: >99% Inf. = 505 mg/L Eff. = 0.5 mg/L MVR	Lead: No influent data. Eff.: <0.001 mg/L Mercury: No influent data. Eff.: <0.005 mg/L Nickel: No influent data. Eff.: 0.002 mg/L Silver: No influent data. Eff.: <0.0002 mg/L Zinc: No influent data. Eff.: 0.02 mg/L Cyanide: No influent data. Eff.: <0.01 mg/L Meets NPDES permit except for TMDLs for hexavalent chromium and mercury	Boron: 97% Inf. = <1 to 3.12 mg/L Eff. = 0.02 to 0.06 mg/L Calcium: 98% Inf. = 1,175 to 1,933 mg/L Eff. = 0.36 to 22.2 mg/L Lithium: 99% Inf. = 9.1 to 14.3 mg/L Eff. = non-detect to 0.13 mg/L Magnesium: 98% Inf. = 109.8 to 176.8 mg/L Eff. = <0.1 to 2.04 mg/L Sodium: 98% Inf. = 4,712 to 7,781 mg/L Eff. = 0.37 to 87.9 mg/L Arsenic: 82% Inf. = <0.01 to 0.028 mg/L Eff. = <0.005 mg/L Titanium: 86% Inf. = <0.01 to 0.037 mg/L Eff. = <0.005 mg/L	

Constituents of concern	Location and results				
	Pinedale Anticline Water Reclamation Facility, Wyoming (Shafer, 2011)	**Maggie Spain Water-Recycling Facility, Barnett Shale, Texas (Hayes et al., 2014)**	**Judsonia, Sunnydale, Arkansas (U.S. EPA, 2015f)**	**9-month study treating Marcellus Shale waste using thermal distillation (Boschee, 2014; Bruff and Jikich, 2011)**	**San Ardo Water Reclamation Facility, San Ardo, California (Conventional oil and gas) (Dahm and Chapman, 2014; Webb et al., 2009)**
Metals, cont.			Settling, biological treatment, induced gas flotation, and MVR	Thermal distillation	
Radio-nuclides	Results not reported.	Results not reported.	Not regulated under permit – believed to be absent.	Radium-226: 97% - 99% Inf. = 130 to 162 pCi/L Eff. = 0.224 to 2.87 pCi/L Radium-228: 97% - 99% Inf. = 45 to 85.5 pCi/L Eff. = 0.259 to 1.32 pCi/L Gross Alpha: 97% - >99% Inf. = 161 to 664 pCi/L Eff. = 0.841 to 6.49 pCi/L Gross Beta: 98% - >99% Inf. = 79.7 to 847 pCi/L Eff. = 0.259 to 1.57 pCi/L Thorium 232: 71% - 90% Inf. = 0.055 to 0.114 pCi/L Eff. = 0.011 to 0.016 pCi/L Thermal distillation	Results not reported.

Constituents of concern	Location and results				
	Pinedale Anticline Water Reclamation Facility, Wyoming (Shafer, 2011)	**Maggie Spain Water-Recycling Facility, Barnett Shale, Texas (Hayes et al., 2014)**	**Judsonia, Sunnydale, Arkansas (U.S. EPA, 2015f)**	**9-month study treating Marcellus Shale waste using thermal distillation (Boschee, 2014; Bruff and Jikich, 2011)**	**San Ardo Water Reclamation Facility, San Ardo, California (Conventional oil and gas) (Dahm and Chapman, 2014; Webb et al., 2009)**
Organics	O&G: 99% Inf. = 50 to 2,400 mg/L Eff. = non-detect BTEX: 99% Inf. = 28 to 80 mg/L Eff. = non-detect Gasoline range organics: RO: 99% Inf. = 88 to 420 mg/L Eff. = non-detect Diesel range organics: 99% Inf. = 77 to 1,100 mg/L Eff. = non-detect Methanol: 99% Inf. = 40 to 1,500 mg/L Eff. = non-detect Oil-water separator, anaerobic and aerobic biological treatment, coagulation, flocculation, flotation, sand filtration, membrane bioreactor, and ultrafiltration	TPH: >80% Inf. = 388 mg/L Eff. = 4.6 mg/L BTEX: 94% Inf. = 3.3 mg/L Eff. = 0.2 mg/L TOC: 48% Inf. = 42 mg/L Eff. = 22 mg/L MVR	Biochemical oxygen demand: No influent data. Eff.: <2 mg/L O&G: No influent data. Eff.: <5 mg/L Benzo (k) fluoranthene: No influent data. Eff.: <0.005 mg/L Bis (2-Ethylhexyl) Phthalate: No influent data. Eff.: <0.001 mg/L Butyl benzyl phthalate: No influent data. Eff.: <0.001 mg/L Meets NPDES permit Settling, biological treatment, induced gas flotation, and MVR	Acetone: 93% Inf. = 8.71 to 13.8 mg/L Eff. = 0.524 to 0.949 mg/L Toluene: >80% Inf. = 0.0083 to 0.0015 mg/L Eff. = non-detect to 0.0013 mg/L Methane: >99% Inf. = 0.748 to 5.49 mg/L Eff. = non-detect to 0.0013 mg/L DRO: 0 to 82% Inf. = 4 to 7.1 mg/L Eff. = 0.99 to 4.9 mg/L O&G: No removal Thermal distillation	Results not reported.

8.5.2.1. Total Suspended Solids

The reduction of TSS is typically required for reuse. Hydraulic fracturing wastewaters containing suspended solids can plug the well and damage equipment if reused for other fracking operations (Tiemann et al., 2014; Hammer and VanBriesen, 2012). For treated water that is discharged, the EPA has a secondary treatment standard for POTWs that limits TSS in the effluent to 30 mg/L (30-day average). In addition, most advanced treatment technologies require the removal of TSS prior to treatment to avoid operational problems such as membrane fouling/scaling and to extend the life of the treatment unit. TSS can be removed by several processes, such as coagulation, flocculation, sedimentation, and filtration (including microfiltration and media and bag and/or cartridge filtration), and with hydrocyclones, dissolved air flotation, freeze-thaw evaporation, electrocoagulation, and biological aerated filters (Boschee, 2014; Igunnu and Chen, 2014; Drewes et al., 2009; Fakhru'l-Razi et al., 2009) (see Appendix F).

Technologies that remove TSS have been employed in the Marcellus Shale (sedimentation and filtration) (Mantell, 2013a); Utica Shale (chemical precipitation and filtration) (Mantell, 2013a); Barnett Shale (chemical precipitation and inclined plate clarifier, >90% removal) (Hayes et al., 2014); and Utah (electrocoagulation, 90% removal) (Halliburton, 2014). Details of examples of operating treatment facilities are provided in Table 8-6.

8.5.2.2. Total Dissolved Solids

The TDS concentration of hydraulic fracturing wastewater is a key treatment consideration, with the TDS removal needed dependent upon the intended use of the treatment effluent. POTW treatment and basic treatment processes at a CWT (i.e., chemical precipitation, sedimentation, and filtration) are not reliable methods for removing TDS. Reduction requires more advanced treatment processes such as RO, nanofiltration, thermal distillation (including MVR), evaporation, and/or crystallization (Olsson et al., 2013; Boschee, 2012; Drewes et al., 2009). RO and thermal distillation processes can treat waste streams with TDS concentrations up to 45,000 mg/L and more than 100,000 mg/L, respectively (Tiemann et al., 2014). As noted in section 8.5.1, pretreatment (e.g., chemical precipitation, flotation, etc.) is typically needed to remove constituents that may cause fouling or scaling or to remove specific constituents not removed by a particular advanced process. Extremely high TDS waters may require a series of advanced treatment processes to remove TDS to desired levels. However, the cost of treating high-TDS waters may preclude facilities from choosing treatment if other options such as deep well injection are available and more cost-effective (Tiemann et al., 2014). Emerging technologies such as membrane distillation and forward osmosis are also showing promise for TDS removal and require less energy compared to other desalination processes (Shaffer et al., 2013).

Examples of facilities with advanced technologies and their effectiveness in reducing TDS concentrations in hydraulic fracturing wastewaters from conventional and unconventional resources are summarized in Table 8-6.

8.5.2.3. Anions

Although chemical precipitation processes can reduce concentrations of multivalent anions such as sulfate, monovalent anions (e.g., bromide and chloride) are not removed by basic treatment processes and require more advanced treatment such as RO, nanofiltration, thermal distillation (including MVR), evaporation, and/or crystallization (Hammer and VanBriesen, 2012).

Ion exchange and adsorption are effective treatment processes for removing fluoride but not typically the anions of concern in hydraulic fracturing wastewaters (bromide, chloride, sulfate) (Drewes et al., 2009). Emerging technologies applicable to TDS will typically remove anions. However, issues discussed above, such as the potential for scaling, still apply.

8.5.2.4. Metals and Metalloids

Chemical precipitation, including lime softening and chemical oxidation, is effective at removing metals (e.g., sodium sulfate reacts with metals to form solid precipitates such as barium sulfate) (Drewes et al., 2009; Fakhru'l-Razi et al., 2009). However, as mentioned in Section 8.5.2.3, chemical precipitation does not adequately remove monovalent ions (e.g., sodium, potassium), and the produced solid residuals from this process typically require further treatment, such as de-watering (Duraisamy et al., 2013; Hammer and VanBriesen, 2012). Media filtration can remove metals if coagulation/oxidation is implemented prior to filtration (Duraisamy et al., 2013). Advanced treatment processes such as distillation (with pH adjustment to prevent scaling), evaporation, RO, and nanofiltration can remove dissolved metals and metalloids (Hayes et al., 2014; Igunnu and Chen, 2014; Bruff and Jikich, 2011; Drewes et al., 2009). However, if metal oxides are present or formed during treatment, they must be removed prior to RO and nanofiltration processes to prevent membrane fouling (Drewes et al., 2009). Also, boron is not easily removed by RO, achieving less than 50% rejection (the percentage of a constituent captured and thus removed by the membrane) at neutral pH (rejection is greater at higher pH values) (Drewes et al., 2009). Ion exchange can be used to remove other metals such as calcium, magnesium, barium, strontium, and certain oxidized heavy metals such as chromate and selenate (Drewes et al., 2009). Adsorption can remove metals but is typically used as a polishing step to prolong the replacement/regeneration of the adsorptive media (Igunnu and Chen, 2014).

The literature provides examples of facilities able to reduce metal and metalloid concentrations in conventional and unconventional hydraulic fracturing wastewaters, some of which are provided in Table 8-6. The facilities in Table 8-6 have achieved removals of 98%–99% for a number of metals. Other work demonstrating effective removal includes a 99% reduction in barium using chemical precipitation (Marcellus Shale region) (Warner et al., 2013a) and over 90% boron removal with RO (at pH of 10.8) at two California facilities (Webb et al., 2009; Kennedy/Jenks Consultants, 2002). However, influent concentration must be considered together with removal efficiency to determine whether effluent quality meets the requirements dictated by end use or by regulations. In the case of the facility described by Kennedy/Jenks Consultants (2002) the boron effluent concentration of 1.9 mg/L (average influent concentration of 16.5 mg/L) was not low enough to meet California's action level of 1 mg/L.

Newer treatment methods for metals removal include electrocoagulation (Halliburton, 2014; Gomes et al., 2009) and electrodialysis (Banasiak and Schäfer, 2009). Testing of electrocoagulation has been performed in the Green River Basin (Halliburton, 2014) and the Eagle Ford Shale (Gomes et al., 2009). While showing promising results in some trials, results of these early studies have illustrated challenges, with removal efficiencies affected by factors such as pH and salt content. Membrane distillation has also shown promise in removing heavy metals and boron in wastewaters (Camacho et al., 2013).

8.5.2.5. Radionuclides

Several processes (e.g., RO, nanofiltration, and thermal distillation) are effective for removing radionuclides (Drewes et al., 2009). Ion exchange can be used to treat for specific radionuclides such as radium (Drewes et al., 2009). Chemical precipitation of radium with barium sulfate has also been shown to be a very effective method for removing radium (Zhang et al., 2014b).

Data on radionuclide removals achieved in active treatment plants are scarce. The literature does provide some data from the Marcellus Shale region on use of distillation and chemical precipitation (co-precipitation of radium with barium sulfate). The nine-month pilot-scale study conducted by Bruff and Jikich (2011) showed that distillation treatment could achieve high removal efficiencies for radionuclides (see Table 8-6), and Warner et al. (2013b) reported that a CWT achieved over 99% removal of radium via co-precipitation of radium with barium sulfate. However, in both studies, radionuclides were detected in effluent samples, and the CWT was discharging to a surface water body during this time (Warner et al., 2013b; Bruff and Jikich, 2011); see Section 8.6.2. Effluent from distillation treatment was found to contain up to 6.49 pCi/L for gross alpha (from 249 pCi/L prior to distillation) (Bruff and Jikich, 2011). Between 2010 and 2012, samples of wastewater effluent from a western PA CWT contained a mean radium level of 4 pCi/L (Warner et al., 2013a).

8.5.2.6. Organics

Because hydraulic fracturing wastewaters can contain various types of organic compounds that each have different properties, specific treatment processes or series of processes are used to target the various classes of organic contaminants. Effectiveness of treatment depends on the specific organic compound and the technology employed (see Appendix F). It should be noted that in many studies, rather than testing for several organic constituents, researchers often measure organics in terms of biochemical oxygen demand and/or chemical oxygen demand, which are an indirect measure of the amount of organic compounds in the water. Organic compounds may also be measured and/or reported in groupings such as total petroleum hydrocarbons (TPH) (which include gasoline range organics (GROs) and diesel range organics (DROs)), oil and grease, VOCs (which include BTEX), and SVOCs.

Based on examples found in literature, facilities have demonstrated the capability to treat for organic compounds in hydraulic fracturing wastewaters using a single process or a series of processes (Hayes et al., 2014; Bruff and Jikich, 2011; Shafer, 2011) (see Table 8-6). The processes can include anaerobic and aerobic biological treatment, coagulation, flocculation, flotation, filtration, bioremediation, ultrafiltration, MVR, and dewvaporation. Forward osmosis is an emerging technology that may be promising for organics removal in hydraulic fracturing wastewaters because it is capable of rejecting the same organic contaminants as commercially-available pressure-driven processes (Drewes et al., 2009).

8.5.2.7. Estimated Treatment Removal Efficiencies

There are relatively few studies that have evaluated the ability of individual treatment processes to remove constituents from hydraulic fracturing wastewater and present the resulting water quality. Furthermore, although a specific technology may demonstrate a high removal percentage for a particular constituent, if the influent concentration of that constituent is extremely high, the constituent concentration in the treated water may still exceed permit limits and/or disposal requirements. Appendix Table F-4 presents the results of simple calculations pairing average hydraulic fracturing wastewater concentrations from Chapter 7 with treatment process removal efficiencies reported in the literature in Table F-2.

As an example, radium in wastewater from the Marcellus Shale and Upper Devonian sandstones can be in the thousands of pCi/L. With a 95% removal rate, chemical precipitation may result in effluent that still exceeds 100 pCi/L. Distillation and reverse osmosis might produce effluent with concentrations in the tens of pCi/L. A radium concentration of 120 pCi/L, however, could be reduced to less than 5 pCi/L by RO or distillation. Wastewater with barium concentrations in the range of 140 – 160 mg/L (e.g., the Cotton Valley and Mesaverde tight sands) might be reduced to concentrations under 5 mg/L by distillation and roughly 11-13 mg/L by RO. Barium concentrations in the thousands of mg/L would be substantially reduced by any of several processes but might still be relatively high and could exceed 100 m g/L. Table F-4 also illustrates the potential for achieving low concentrations of organic compounds in wastewater treated with freeze-thaw evaporation or advanced oxidation and precipitation.

This analysis is intended to highlight the potential impacts of influent concentration on treatment outcome and to illustrate the relative capabilities of various treatment processes for an example set of constituents. Removal efficiencies would differ and likely be greater with a full set of pretreatment and treatment processes that would be seen in a CWT (see Table 8-6).

8.5.3. Design of Treatment Trains for CWTs

Based on the chemical composition of the hydraulic fracturing wastewater and the desired effluent water quality, a series of treatment technologies will most likely be necessary. The possible combinations of unit processes to formulate treatment trains are extensive. One report identified 41 different treatment unit processes that have been used in the treatment of oil and gas wastewater and 19 unique treatment trains (combinations of unit processes) (Drewes et al., 2009). Fakhru'l-Razi et al. (2009) also provide examples of process flow diagrams that have been used in pilot-scale and commercial applications for treating oil and gas wastewater. Figure 8-8 shows the treatment train for the Pinedale Anticline Facility, which includes pretreatment for dispersed oil, VOCs, and heavy metals and advanced treatment for removal of TDS, dissolved organics, and boron. This CWT can either discharge to surface water or provide the treated wastewater to operators for reuse.

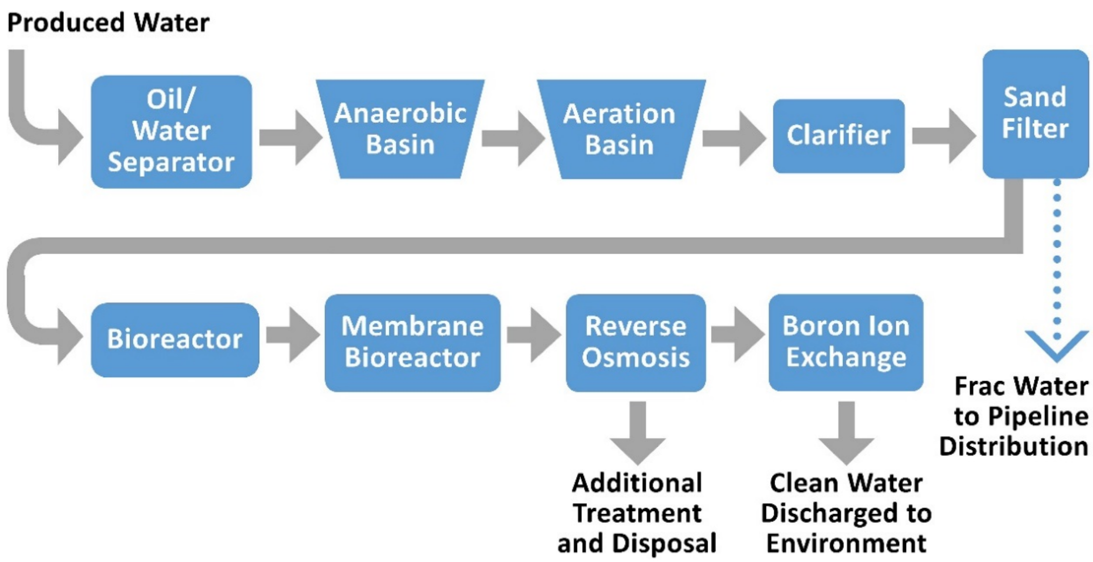

Figure 8-8. Full discharge water process used in the Pinedale Anticline field.

Source: Boschee (2012).

Table 8-7 provides information on some CWTs in locations across the country and the processes they employ. The table also notes for each facility whether data are readily available on effluent quality. Comprehensive and systematic data on influent and effluent quality from a range of CWTs that treat to a variety of water quality levels is difficult to procure, rendering it challenging to understand removal efficiencies and resulting effluent quality, especially when a facility offers a range of water quality products (e.g., for reuse vs. discharge). For those facilities with NPDES permits, discharge monitoring report (DMR) data may be available for some constituents, although if the facility does not discharge regularly, these data will be sporadic.

CWTs such as the Judsonia Central Water Treatment Facility in Arkansas, the Casella-Altela Regional Environmental Services, and Clarion Altela Environmental Services (see Table 8-7) facilities have NPDES permits and use MVR or thermal distillation for TDS removal. As of March 2015, the Pinedale Anticline Facility and the Judsonia Facility appear to be the only CWTs in Table 8-7 discharging to a surface water body.

Table 8-7. Examples of centralized waste treatment facilities.

Facility	State	Description of Unit Processes	Does CWT have a NPDES permit for discharge?	Does CWT provide effluent for reuse?	Does CWT have advanced process for TDS removal?	What is the status of the facility as of January 2015?	Are effluent quality data available through literature search?
Pinedale Anticline Water Reclamation Facility[a]	WY	Oil/water separation, biological treatment, aeration, clarification, sand filtration, bioreactor, membrane bioreactor, RO, and ion exchange	No - However, facility is permitted to discharge under 40 CFR 435 Subpart E (WY0054224). Facility is permitted to discharge up to 25% of its effluent stream	Yes	Yes, RO (Boschee, 2014, 2012)	The treatment plant produces treated water for reuse and for discharge to surface water. The website indicates the facility is in operation and is recycling to support drilling operations and is discharging to the New Fork River (http://hswater.squarespace.com/pinedale-anticline/).	Yes – DMR data available on Wyoming DEQ website. Some information can also be obtained from Shafer (2011).
SEECO – Judsonia Water Reuse Recycling Facility	AR	Settling, biological treatment, induced gas flotation, and MVR	Yes - AR0052051	Yes	Yes, MVR	The treatment plant provides treated water for reuse and for discharge to surface water. Based on DMR data from late 2014-early 2015, the system is discharging treated water to a surface water body, though intermittently.	DMR data available

Facility	State	Description of Unit Processes	Does CWT have a NPDES permit for discharge?	Does CWT provide effluent for reuse?	Does CWT have advanced process for TDS removal?	What is the status of the facility as of January 2015?	Are effluent quality data available through literature search?
Eureka Resources – Williamsport 2nd Street Facility	PA	Settling, oil/water separation, chemical precipitation, clarification, MVR. Can treat with or without TDS removal.	No - However, future plans to install RO for direct discharge capability	Yes	Yes, MVR	Per Ertel et al. (2013), the facility provides treatment wastewater for reuse and indirect discharge. The facility treats entirely or almost entirely hydraulic fracturing wastewater.	No
Standing Stone Facility, Bradford County	PA	Settling, oil/water separation, chemical precipitation, clarification, MVR, crystallization	Yes - PA0232351	Yes	Yes, MVR, crystallizer	The facility can provide treated wastewater for reuse and also has received an NPDES permit for direct discharge. The facility treats hydraulic fracturing wastewater.	No

Facility	State	Description of Unit Processes	Does CWT have a NPDES permit for discharge?	Does CWT provide effluent for reuse?	Does CWT have advanced process for TDS removal?	What is the status of the facility as of January 2015?	Are effluent quality data available through literature search?
Wellington Water Works	CO	Dissolved air flotation, pre-filtration, microfiltration with ceramic membranes, activated carbon adsorption. Water is pumped to an aquifer storage and recovery well. Water is then extracted and treated with RO (Alzahrani et al., 2013).	Permit number issued by CO (61879)	Yes	Yes, RO but only after the water is sent to an aquifer storage and recovery well	Per Stewart (2013b), the facility is providing treated wastewater for reuse, for agricultural use, to a shallow well to augment the municipal drinking water supply, and for discharge to the Colorado River.	No
Casella Altela Regional Environmental Services (CARES) McKean Facility	McKean County, PA	Pretreatment system (not defined in literature) and thermal distillation	Yes – PA0102288	Yes	Yes – thermal distillation	The treatment plant is capable of reuse and recycle for fracturing operations and surface water discharge of excess water. However, the facility's website indicates it is only treating water for reuse/recycle as of early 2015 (http://caresforwater.com/location/cares-mckean).	No - just NPDES discharge requirements

Facility	State	Description of Unit Processes	Does CWT have a NPDES permit for discharge?	Does CWT provide effluent for reuse?	Does CWT have advanced process for TDS removal?	What is the status of the facility as of January 2015?	Are effluent quality data available through literature search?
Clarion Altela Environmental Services (CAES) Facility	Clarion County, PA	Pretreatment system (not defined in literature) and thermal distillation	Yes – PA0103632	Yes	Yes – thermal distillation	The treatment plant capable of reuse and recycle for fracturing operations and surface water discharge of excess water. However, the facility's website indicates it is only treating water for reuse/recycle as of early 2015 (http://caeswater.com/technology/).	No – just NPDES discharge requirements
Terraqua Resource Management (aka. Water Tower Square Gas Well Wastewater Processing Facility)	Lycoming County, PA	Equalization tanks, oil-water separation via chemical addition (sulfuric acid, emulsion breaker), pH adjustment, coagulation, flocculation, inclined plate clarifier, sand filtration	Yes – PA0233650 Permit pending approval for discharge to stream (as of 4/17/2009)	Yes	No – However, TARM recognizes that they can't discharge, until they install TDS treatment	According to its website (last updated 2012), the facility reuses/recycles treated water for fracturing operations (http://www.tarmsolutions.com/solutions/).	No
Maggie Spain Water-Recycling Facility	Decatur, TX	Settling, flash mixer with lime and polymer addition, inclined plate clarifier, surge tank, MVR	No	Yes	Yes – MVR	A 17-month pilot study using a commercial-scale mobile treatment facility was concluded in 2011. The status is unclear as of early 2015.	Yes – Some information can be obtained from Hayes et al. (2014).

Facility	State	Description of Unit Processes	Does CWT have a NPDES permit for discharge?	Does CWT provide effluent for reuse?	Does CWT have advanced process for TDS removal?	What is the status of the facility as of January 2015?	Are effluent quality data available through literature search?
Fountain Quail/NAC Services - Kenedy	Kenedy, TX	Oil-water separator, coagulation, flocculation, sedimentation, filtration, MVR.	No	Yes	Yes – MVR	According to its website, the facility reuses/recycles treated water for fracturing operations (http://www.aqua-pure.com/operations/shale/ford/ford.html).	No
Purestream - Gonzales facility	Gonzales, TX	Induced gas flotation and MVR	No	Yes	Yes - MVR	Per Dahm and Chapman (2014) commercial operations deployed March 2014 for reuse/recycle for fracturing operations.	No
LINN Energy Fyre Ranch - Granite Wash	Wheeler County, TX	Induced gas flotation and MVR	No	Yes	Yes - MVR	AVARA system installed for reuse/recycle in June 2014. http://purestream.com/index.php/water-management/vapor-recompression/photos-and-videos	No

Facility	State	Description of Unit Processes	Does CWT have a NPDES permit for discharge?	Does CWT provide effluent for reuse?	Does CWT have advanced process for TDS removal?	What is the status of the facility as of January 2015?	Are effluent quality data available through literature search?
Fluid Recovery Service Josephine Facility[b]	PA	Oil-water separator, aeration, chemical precipitation with sodium sulfate, lime, and a polymer, inclined plate clarifier	Expired - PA0095273	No	No	The facility claims to have stopped accepting Marcellus wastewater September 30, 2011 (Ferrar et al., 2013). It treats conventional oil and gas wastewater. The facility will be upgrading to include evaporative technology that will enable it to attain monthly average TDS levels of 500 mg/L or less.	Yes – Some effluent results obtained from Ferrar et al. (2013) and Warner et al. (2013a). Also minimal DMR data from the EPA.
Fluid Recovery Service Franklin Facility[b]	PA	Oil-water separator, aeration, chemical precipitation with sodium sulfate, lime, and a polymer, inclined plate clarifier	Expired - PA0101508	No	No	This facility is not accepting wastewater from hydraulic fracturing operations as of January 2015. The facility will be upgrading to include evaporative technology that will enable it to attain monthly average TDS levels of 500 mg/L or less.	Minimal DMR data from the EPA.

Facility	State	Description of Unit Processes	Does CWT have a NPDES permit for discharge?	Does CWT provide effluent for reuse?	Does CWT have advanced process for TDS removal?	What is the status of the facility as of January 2015?	Are effluent quality data available through literature search?
Hart Resources-Creekside Facility[b]	PA	Oil-water separator, aeration, chemical precipitation with sodium sulfate, lime, and a polymer, inclined plate clarifier	Expired - PA0095443	No	No	This facility is not accepting wastewater from hydraulic fracturing operations as of January 2015. The facility will be upgrading to include evaporative technology that will enable it to attain monthly average TDS levels of 500 mg/L or less.	Minimal DMR data from the EPA.

[a] For Pinedale Anticline Water Reclamation Facility, surface water discharges are permitted under 40 CFR 435 Subpart E (beneficial use subcategory agricultural and wildlife water) not 40 CFR 437 (the discharge permit for CWTs). For the purposes of this assessment, this facility is included with CWTs.

[b] As of May 15, 2013, these facilities are under an administrative order (AO). According to the AO, these facilities must comply with a monthly effluent limit for TDS not to exceed 500 mg/L. This will allow them to treat high-saline wastewaters typical of unconventional oil and gas operations. To meet the requirements of the AO, they have applied to PADEP for a NPDES permit and are planning to install treatment for TDS.

8.6. Potential Impacts on Drinking Water Resources

Several articles have noted potential effects of hydraulic fracturing wastewater on water resources (Vengosh et al., 2014; Olmstead et al., 2013; Rahm et al., 2013; States et al., 2013; Vidic et al., 2013; Rozell and Reaven, 2012; Entrekin et al., 2011), with one study using probability modeling indicating that water pollution risk associated with gas extraction in the Marcellus Shale is highest for the wastewater disposal aspects of the operation (Rozell and Reaven, 2012). Whether drinking water resources are affected by hydraulic fracturing wastewater depends at least in part upon the characteristics of the wastewater, the form of discharge or other management practice, and the processes used if the wastewater is treated. Other site-specific factors (e.g., size of receiving water and volume of wastewater) determine the magnitude and nature of potential effects, but a thorough exploration of local factors is beyond the scope of this assessment. The majority of hydraulic fracturing wastewater is either injected into a disposal well or, in the case of the Marcellus region, reused for other hydraulic fracturing jobs. Potential impacts on drinking water resources may occur on a local level through several routes: treated wastewater may be discharged directly from centralized waste treatment facilities (CWTs) or indirectly from publicly owned treatment works (POTWs) that receive CWT effluent; sediments in water bodies receiving effluent may accumulate contaminants; spills or leaks may be associated with on-site storage or transportation (see Chapter 7); and in previous years, hydraulic fracturing wastewater treated at POTWs was discharged to surface waters.

It has been suggested that the most significant effects of hydraulic fracturing on surface water quality are related to discharges of partially treated wastewater, although these effects vary according to region (Kuwayama et al., 2015). A recent study (Bowen et al., 2015) concluded that there is currently no clear evidence of national-level trends in surface water quality (as measured by specific conductivity and chloride) in areas where unconventional oil and gas production is taking place. These authors note that available national level databases have limitations for assessing this question.

Pits and impoundments associated with waste management may have impacts on drinking water resources and are discussed in Chapter 7. In addition, unauthorized discharge of wastewater is a potential mechanism for impacts on drinking water resources. Descriptions of several incidents and resulting legal actions have been publicly reported. However, such events are not generally described in the scientific literature, and the prevalence of this type of activity is unclear.

Important considerations regarding the potential impact of hydraulic fracturing wastewater on a receiving waterbody include whether constituents in the wastewater are known to have health effects, if they are regulated drinking water contaminants, or if they may give rise to regulated compounds. For some classes of constituents, such as disinfection by-product (DBP) precursors, considerable research exists. For others, information is limited regarding their concentrations in effluents and whether they are likely to affect drinking water at intakes. The following subsections identify several classes of constituents known to occur in hydraulic fracturing wastewater, discuss whether potential impacts are likely, and provide specific examples of information gaps.

8.6.1. Bromide and Chloride

Bromide and chloride are two constituents commonly found in high-total dissolved solids (TDS) hydraulic fracturing wastewater. As noted in section 8.3.1.1, chloride is a regulated contaminant with a secondary MCL standard of 250 mg/L. Bromide is not regulated but is of concern due to its role in the formation of DBPs (Parker et al., 2014; Krasner, 2009) (see Appendix F for information on DBP formation). High-TDS wastewaters from the Marcellus Shale can be of concern because the limited availability of underground injection for disposal can result in a higher rate of discharge of treated wastewater to surface waters compared to other parts of the country. In response to concerns in part over bromide in discharges, operators in Pennsylvania have discontinued the practice of sending wastewater from hydraulic fracturing operations to POTWs (States et al., 2013). Also, CWTs have been shifting towards treatment of those wastewaters for reuse rather than discharging to surface water bodies (Hammer and VanBriesen, 2012).

States et al. (2013) found a strong correlation between bromide concentrations in source water from the Allegheny River in Pennsylvania and the percentage of brominated trihalomethanes in finished drinking water. The authors noted that source water containing 50 μg/L bromide resulted in treated water with approximately 62% of its finished water total trihalomethanes consisting of bromoform, dibromochloromethane, and bromodichloromethane. Source water containing 150 μg/L bromide resulted in finished water TTHMs composed of approximately 83% brominated species. Allegheny River bromide concentrations measured during the study ranged from less than 25 μg/L to 299 μg/L, with the highest bromide concentrations measured under low-flow conditions. Industrial wastewater sites accounted for approximately 50% of the increase in bromide load as water moved downriver.

In addition, a related constituent, iodide can be a constituent in hydraulic fracturing wastewater (see Chapter 7). Although its effects have not been as well documented as those associated with bromide (Xu et al., 2008), iodide raises some of the same concerns (such as DBP formation) as bromide does (Parker et al., 2014; Krasner, 2009). Iodinated DBPs are not regulated by the EPA as of early 2015.

As discussed in Section 8.5, removal of dissolved solids, including chloride and bromide, requires advanced treatment processes such as reverse osmosis (RO), distillation, evaporation, or crystallization. Unless the treatment plant receiving the high-TDS wastewater employs processes specifically designed to remove these constituents, effluent discharge may contain high levels of bromide and chloride. Drinking water treatment plants with intakes downstream of these discharges may receive water with correspondingly higher levels of bromide and chloride and may have difficulty complying with Safe Drinking Water Act (SDWA) regulations related to DBPs.

8.6.1.1. Effects on Receiving Streams

Studies show that discharges from oil and gas wastewater treatment facilities can elevate TDS, bromide, and chloride levels in receiving waters (States et al., 2013; Wilson and Van Briesen, 2013). Wilson and Van Briesen (2013) measured bromide, chloride, and other constituents at water intakes downstream of wastewater discharges for three years in the Monongahela River in western Pennsylvania. By evaluating water chemistry data in the context of flow measurements, the authors attributed an overall decrease in bromide concentrations from 2010 to 2012 to a decrease in bromide loading; they note that this is likely to be associated with a decrease in management of fossil fuel wastewater at treatment plants that discharge to surface water.

Although treatment plant effluents will be diluted upon reaching the receiving water, the dilution may not be adequate to avoid water quality problems if there are existing pollutant loads in the waterbody from contributors such as acid mine drainage or power plant effluents (Ferrar et al., 2013). Warner et al. (2013a) evaluated effluent from the Josephine Brine Treatment Facility (which treated both conventional and unconventional oil and gas wastewater at the time of the study) and concluded that even a 500 to 3,000-fold dilution of the wastewater would not reduce bromide levels to background. In addition, downstream levels of chloride in the receiving stream were elevated, with a downstream value of 88 mg/L as compared to an upstream value of 18 mg/L.

A study by Hladik et al. (2014) focused on sampling at sites downstream and near the outfalls of plants that treated oil and gas wastewater in Pennsylvania. The authors documented brominated and iodinated DBPs (e.g., dibromochloronitromethane (DBNM); dibromoiodomethane) at the outfalls of CWTs and POTWs and noted that this DBP signature was different than for those plants that did not accept oil and gas wastewater. For example, concentrations of dibromochloronitromethane ranged from 0.26 to 8.7 μg/L, and dibromoiodomethane was measured at 0.98 and 1.3 μg/L; neither compound was detected at an upstream site or at the outfall of the POTW not accepting oil and gas wastewater. These brominated and iodinated compounds are considered more toxic than other types of DBPs (Richardson et al., 2007). Hladik et al. note that these elevated DBP levels could contribute to DBPs at downstream drinking water intakes and can also be an indicator of the potential for more highly brominated and iodinated DPBs forming in drinking treatment plants downstream of these discharges (Hladik et al., 2014). The sites studied by Hladik et al. (2014) received wastewater from both conventional and unconventional oil and gas development.

Research suggests that a relatively small portion of hydraulic fracturing wastewater effluent can notably affect DBP formation. In laboratory studies, Parker et al. (2014) diluted hydraulic fracturing wastewater from the Marcellus and Fayetteville shales with Allegheny and Ohio River waters and then disinfected the mixtures. In chlorinated samples containing as little as 0.01% hydraulic fracturing wastewater, the THM composition shifted significantly away from chloroform species to a greater representation of brominated and iodinated species.

Elevated concentrations of bromide in effluents can place a burden on downstream drinking water treatment systems. States et al. (2013) studied influent and finished water at the Pittsburgh Water and Sewer Authority (PWSA) drinking water system, concluding that elevated bromide in the source water led to elevated total trihalomethanes (TTHM) formation in the treated drinking water. The authors also noted a substantial increase in the percentage of brominated TTHMs (States et al., 2013), as discussed above. The utility modified their treatment process and proposed improvements to their storage facilities to address the elevated TTHM levels in the distribution system (Chester Engineers, 2012).

8.6.1.2. Modeling

The EPA's contaminant modeling shows that that the strategies most likely to reduce bromide impacts on downstream users include reducing effluent concentrations (e.g., discharging flowback versus produced water), discharging during higher stream flow periods, and using a pulsing or intermittent discharge. Weaver et al. (In Press) developed a computer model to estimate river and stream bromide concentrations after treated water discharges. The model utilizes existing data for bromide concentrations in produced water, flowback, and mixtures, combined with existing stream flow data from USGS stations in Pennsylvania. The model parameters include steady state versus transient inputs to receiving waters, high and low streamflow months, varying effluent concentration and types (produced, flowback, and mixed). For steady-state scenarios in the model, bromide concentrations are lowest under high flow conditions with lower concentrations of effluent (flowback and mixed water).

A source apportionment study conducted by the EPA considered the relative contributions of bromide, chloride, nitrate, and sulfate from CWTs primarily treating hydraulic fracturing wastewater to the Allegheny River Basin and to water at two downstream public water system intakes on the Allegheny River (U.S. EPA, 2015p). The Allegheny River and its tributaries receive runoff and discharges containing an array of contaminants, including these anions. Contaminant sources include discharges from CWTs for oil and gas wastewater, runoff from acid mine drainage and mining operations, discharges from coal-fired electric power stations, industrial wastewater treatment plant effluents, and POTW discharges. The Allegheny River is also the water supply for thirteen public water systems that serve over 500,000 people in western Pennsylvania.

In Pennsylvania, wastewater produced from hydraulic fracturing of the Marcellus formation has been mostly diverted from CWTs and POTWs that discharge to public waters in the state (Hammer and VanBriesen, 2012). Wastewater produced from hydraulic fracturing of non-Marcellus formations, however, continues to be sent to surface-discharging facilities on the Allegheny River.

The source apportionment study considered contributions of bromide, chloride, nitrate and sulfate to public water supplies from CWTs and other upriver sources by: developing chemical source profiles, or fingerprints, for discharges upstream of the public water system intakes; characterizing water quality in the river upstream and downstream of the CWTs, electric generating stations, and industrial facilities; characterizing the water quality at the public water system intakes; and analyzing the sampling data collected with the EPA Positive Matrix Factorization (PMF) receptor model in order to quantify relative contributions of contaminant sources to the anions found at the public water system intakes. The study focused on low-flow conditions.

CWTs and coal-fired power plants with flue gas desulfurization were found to contribute bromide to the two public water supply intakes. Although acid mine drainage also contributed bromide, its contribution was minor (9% at one intake) compared to the contributions from the CWTs (89% and 37% at the two intakes) and coal-fired power plants (50-59% at one intake). The CWTs, coal-fired power plants, and acid mine drainage combined accounted for 88–89% of the bromide at one intake and 96% of the bromide at the other intake.

8.6.1.3. Summary

Most drinking water treatment plants are not designed to address high concentrations of TDS (including bromide and iodide), limiting their options for restricting the formation of brominated and iodinated DBPs. Tighter restrictions on TDS in effluent from POTWs and CWTs have led to a reduction in in-stream bromide concentrations. Advanced treatment processes at CWTs such as reverse osmosis, distillation, evaporation, or crystallization can reduce chloride, bromide, and iodide in surface waters. Strategies such as reducing effluent concentrations, discharging during higher stream flow periods, and utilizing a pulsing or intermittent discharge could also reduce the potential impact of elevated TDS on drinking water treatment plants.

8.6.2. Radionuclides

Potential impacts on drinking water resources from technologically enhanced naturally occurring radioactive material (TENORMs) associated with hydraulic fracturing wastewater may arise from a number of sources, including: treated wastewater that does not have adequately reduced radionuclide concentrations, accumulation of radionuclides in surface water sediments downstream of wastewater treatment plant discharge points, migration from soils that have accumulated radionuclides from previous activities such as pits or land application, and inadequate management of treatment plant solids that have accumulated radionuclides (such as filter cake).

In Pennsylvania between 2007 and 2010, TENORM-bearing produced wastewaters were sent to POTWs, which are generally not required to monitor for radioactivity (Resnikoff et al., 2010). Although the practice of management of Marcellus waters via POTWs has declined, there is still potential for input of radionuclides to surface waters via discharge of CWT effluents either directly to surface waters or indirectly through discharge to POTWs.

Data regarding TENORM content in oil and gas wastes that are treated and discharged to surface waters are limited. However, a recent study by the Pennsylvania Department of Environmental Protection (PA DEP) (PA DEP, 2015b) provides information that helps fill this data gap. The study began in 2013 and examined radionuclide (radium-226, radium-228, K-40, gross alpha, gross beta) levels at 29 wastewater plants in Pennsylvania that cover a range of both sources and treatment plant types, including POTWs, CWTs that treat oil and gas wastewaters and can discharge to surface water or a POTW, and zero liquid discharge facilities treating oil and gas wastewater. Four of the 10 discharging CWTs sampled during the study discharged to surface water under a National Pollution Discharge Elimination System (NDPES) permit, and the others discharged to POTWs. Six of the POTWs in the study received effluent from a CWT along with municipal wastewater. The CWTs in the study are not described as receiving exclusively Marcellus wastewater, but the study itself was motivated by concerns over an increase in radionuclides in oil and gas wastes observed during the expansion of Marcellus Shale production.

The POTWs receiving influent from CWTs treating oil and gas wastewater (along with municipal wastewater influent) had average effluent radium-226 concentrations of 103 pCi/L (unfiltered) and 129 pCi/L (filtered) (filtration is used to remove very fine particulates from the water). Those POTWs not receiving influent from CWTs treating oil and gas wastewater effluent had higher average radium-226 values in unfiltered samples (145 pCi/L) and lower values for filtered samples

(47 pCi/L). For perspective, the maximum contaminant level (MCL) for radium-226 plus radium-228 is 5 pCi/L. For reference, radium-226 in river water generally ranges from 0.014 pCi/L to 0.54 pCi/L (0.5 to 20 mBq/L) (IAEA, 2014). The results of the POTW sampling are inconclusive as to whether the effluents from POTWs receiving CWT-treated oil and gas wastewater are routinely higher than the effluents from those without this type of influent.

For the CWTs in the PA DEP study, average radium-226 content in the effluents was an order of magnitude higher than in effluents from the POTWs (1,840 pCi/L unfiltered, 2,100 pCi/L, filtered). The effluent averages were similar to averages for the influent concentrations, although median concentrations in the effluents were much lower than in the influents. Effluent from zero-discharge facilities averaged 2,610 pCi/L radium-226 and 295 pCi/L radium-228, although these effluents would most likely be reused as fracturing fluid (PA DEP, 2015b). The authors do note a potential for environmental effects from spills of influent or effluent from zero-discharge facilities.

Warner et al. (2013a) noted that if the activities of radium-226 and radium-228 in Marcellus brine influent at the CWT they studied are similar to those reported by other researchers (Rowan et al., 2011), then the CWT achieved a 1,000-fold reduction in radium content via a process of radium coprecipitation with barium sulfate. The detection of radium in effluents from this CWT (mean values of 4 pCi/L of radium-226 and 2 pCi/L of radium-288) even with what may be high treatment removal efficiency underscores the fact that effluent concentrations depend not only upon the treatment processes used but also the influent concentration.

An additional concern related to evaluation of radionuclide concentrations in wastewater is that the high TDS content of hydraulic fracturing wastewater can result in poor recovery of chemical constituents when using wet chemical techniques, leading to underestimations of constituent concentrations. In particular, recovery for radium may be as low as <1% (Nelson et al., 2014). Underestimation of radium content may lead to failure in identifying an impact or potential impact on drinking water resources.

In addition to concerns over of the potential for TENORM in discharges to surface waters, there are may be a legacy of accumulation of radionuclides in surface water sediments. Studies of effluent, stream water, and stream sediment associated with a CWT in western Pennsylvania that has treated both conventional and unconventional oil and gas wastewaters indicate that radium-226 levels in stream sediment samples at the point of discharge are approximately 200 times greater than upstream and background sediments. This indicates the potential for accumulation of contaminants in localized areas of wastewater discharge facilities (Warner et al., 2013a). Although the CWT in question also accepted conventional oil and gas wastewater, Warner et al. (2013a) observed that the radium-228/radium-226 ratio in the river sediments near the discharge (0.22-0.27) is consistent with ratios in Marcellus wastewater. The authors interpret this as an indication that the radium accumulated in the sediments originated from the discharge of treated unconventional oil and gas wastewater. Another study, however, did not find elevated levels of alkali earth metals (including radium) in sediments just downstream of the discharge points of five POTWs that had previously treated Marcellus wastewater (Skalak et al., 2014). Accumulation of contaminants in sediment may be dependent on treatment processes and their removal rates for various constituents as well as stream chemistry and hydrologic characteristics. Contamination with radium-226 would be potentially be long lived; the half-life of radium-226 is approximately 1,600 years, while the half-life of radium-228 is 5.76 years.

The recent PA DEP study (PA DEP, 2015b) found that the radium-226 content in sediments near the discharge points for POTWs receiving treated oil and gas effluent from CWTs (along with their municipal wastewater influent) exceeds typical background soil levels of approximately 1 to 2 pCi/g of radium-226 and radium-228. The authors conclude that wastewater effluent is the most likely source for the radium in these samples. Results indicate an average of 9.00 pCi/g radium-226 and 3.52 pCi/g radium-228 in sediments near outfalls of POTWs. Sediments at 4 CWTs receiving oil and gas wastewater and that discharge to surface water have much higher average concentrations of 84.2 pCi/g for radium-226 and 19.8 pCi/g for radium-228. However, the concentrations of radium in the sediments does not correlate with concentrations of radium in the effluents suggesting that sorption over time affects the concentration of radium in the sediments (PA DEP, 2015b).

The association of radium with sediments near discharge points is attributed to adsorption of radium to the sediments, a process governed by factors such as the salinity of the water and sediment characteristics. In particular, radium has a high affinity for iron and manganese (hydr)oxides in sediment. Increased salinity promotes desorption of radium from sediments, while lower salinity promotes adsorption, with radium adsorbing particularly strongly to sediments high in iron and manganese (hydro)oxides (Porcelli et al., 2014; Gonneea et al., 2008). Warner et al. (2013a) speculate that the discharge of saline CWT effluent into less saline stream water facilitates sorption of radium onto streambed sediments. The long-term fate of radium sorbed to sediments depends upon changes in water salinity and the sediment properties, including any redox processes that may affect iron and manganese minerals in the sediments.

Other solids may contain radionuclides; filter cake samples from treatment at POTWs were found by PA DEP (2015b) to have radium contents greater than typical soil concentrations, and they exhibited a large variation. Filter cake from CWTs had radium concentrations higher than in POTW filter cake. The authors conclude that although the risk to workers and the public from handling and temporary storage of these materials is minimal, there may be environmental risks from spills or long term disposal. There could be impacts on surface waters through spills or effects on ground waters from landfill leachate.

Radionuclide accumulation in CWTs or POTWs may continue to affect the plant even after discontinuing treatment of high radionuclide wastewater. Radium can adsorb onto scales in pipes and tanks and will also co-precipitate calcium, barium, and strontium in sulfate minerals (USGS, 2014e). Pipe scale in oil and gas production facilities can have radium concentrations as high as 154,000 pCi/g, although concentrations of less than 13,500 pCi/g are more common (Schubert et al., 2014). A similar issue, the potential for accumulation and possible release of radionuclides and other trace inorganic constituents in water distribution systems has gained attention, with the potential for drinking water concentrations to exceed drinking water standards (Water Research Foundation, 2010). Scale eventually removed from pipes or other equipment may end up in landfills and then leach into groundwater or run off to a surface water body (USGS, 2013c). Although barium sulfate phases are relatively insoluble, one study demonstrates that barium sulfate scales that were buried in soil could be reduced by microbially mediated processes, allowing release of co-precipitated elements such as radium due to leaching by rainwater (Swann et al., 2004). Monitoring would be needed in order to ascertain the potential for accumulation and release of radionuclides from systems that have treated or continue to treat hydraulic fracturing wastewaters with elevated TENORM concentrations.

Accumulation of radionuclides (potassium, thorium, bismuth, radium, and lead) has been evaluated in two pits in Texas that have stored fluids associated with hydraulic fracturing (Rich and Crosby, 2013). Gamma radiation in these pits has been found to vary from 8 to 23 pCi/g, with beta radiation varying from 6 to 1329 pCi/g (Rich and Crosby, 2013). Although the study sample size was small, the results suggest that radionuclides associated with sediments from some reserve pits could have potential impacts on surface waters or ground waters. This could happen through migration of affected sediments or soils to surface waters or through leaching to ground water.

Salt and radionuclide accumulation can occur near road spreading sites; one study in Pennsylvania found a 20% increase in radium concentrations in soils near roads where wastewaters from conventional operations had been spread for de-icing (Skalak et al., 2014). Accumulation of radionuclides in soils near roads presents a vehicle for potential impacts on drinking water resources. The extent to which hydraulic fracturing wastewater contributes to this depends upon state-level regulations regarding whether hydraulic fracturing wastewater can be used for road spreading.

Effluents and receiving waters can be monitored for radionuclides. Research suggests that radium-226 and radium-228 are the predominant radionuclides in Marcellus Shale wastewater, and they account for most of the gross alpha and gross beta activity in the waters studied (Rowan et al., 2011). Gross alpha and gross beta measurement may therefore serve as an effective screening mechanism for overall radionuclide concentrations in hydraulic fracturing wastewater. This in turn can help in evaluating management strategies. Portable gamma spectrometers allow rapid screening of wastewater effluents. Sediments may also be measured for radionuclide concentrations at discharge points.

8.6.3. Metals

Given the presence in hydraulic fracturing wastewaters of some heavy metals, as well as barium and strontium concentrations that can reach hundreds or even thousands of mg/L (see Table 7-10), surface waters may be impacted if discharges from CTWs or POTWs indirectly receiving oil and gas wastewater via CWTs are not managed appropriately. Spills may also affect surface waters.

Common treatment processes, such as coagulation, are effective at removing many metals (see Section 8.5.2.4). A request by the EPA for effluent sampling from seven facilities in Pennsylvania treating oil and gas wastewaters revealed low to modest concentrations of copper (0-50 μg/L), zinc (14 – 256 μg/L), and nickel (8 – 22 μg/L) (U.S. EPA, 2015d, e). However, metals such as barium and strontium have been found to range from low to elevated in some CWT effluents. For the year 2011, for example, effluent from a Pennsylvania CWT had average barium levels ranging from 9 to 98 mg/L (PA DEP, 2015a). That facility was operating with a barium removal stage and was treating both conventional and hydraulic fracturing wastewater, although effluent concentrations dropped after May, 2011. The facility is scheduled to upgrade its TDS removal capabilities.

Data collected by the EPA between October 2011 and February 2013 at seven Pennsylvania facilities indicate effluent barium concentrations ranging from 0.35 to 25 mg/L (median of 3.5 mg/L and average of 6.7 mg/L). Strontium concentrations ranged from 0.36 to 546 mg/L (median of 297 mg/L and mean of 236 mg/L (U.S. EPA, 2015e). A December 2010 effluent sampling effort in at a discharging CWT in Pennsylvania reported average barium and strontium concentrations of 27 mg/L and nearly 3,000 mg/L, respectively (eight samples from one plant) (Volz et al., 2011). The facility treats conventional oil and gas wastewaters, and it also received Marcellus wastewater until September, 2011.

Limited data are available on metal concentrations in wastewater and treated effluents that are directly discharged; additional information would be needed to assess whether there will be downstream effects on drinking water utilities. NPDES discharge permits, which restrict TDS discharge concentrations, will likely reduce metal effluent concentrations due to the additional treatment necessary to minimize TDS.

8.6.4. Volatile Organic Compounds

Benzene is a common constituent in hydraulic fracturing wastewater, and it is of concern due to recognized human health effects. A wide range of concentrations of BTEX compounds occurs in wastewater from the Barnett and Marcellus shales. Natural gas formations generally produce more BTEX than oil formations (Veil et al., 2004). Generally, lower concentrations of BTEX occur in wastewater from coalbed methane (CBM) production (see Appendix Table E-9). Processes such as aeration or dissolved air flotation can remove volatile organic compounds (VOCs) during treatment, but if treatment is not adequate, the VOCs may reach water resources. The average concentration of benzene in a December 2010 sampling effort was 12 µg/L in the discharge of a Pennsylvania CWT (Volz et al., 2011). The facility was receiving wastewater from both conventional and unconventional operations at that time. Ferrar et al. (2013) measured mean concentrations of benzene, toluene, ethylbenzene, and xylene in effluents from the same facility, and mean concentrations among the four compounds ranged from about 2 to 46 µg/L. Concentrations were lower for samples taken after May 19, 2011 than before, and the effect was considered statistically significant. The treatment processes at this facility do not include aeration.

Leakage from pits or spills creates another potential route of entry to drinking water resources. VOCs have been measured in groundwater near the Duncan Oil Field in New Mexico, downgradient of an unlined pit storing oil and gas wastewater (Sumi, 2004; Eiceman, 1986). VOCs and oil were also found in groundwater about 213 feet (65m) downgradient from an unlined pit in Oklahoma (Kharaka et al., 2002).

8.6.5. Semi-Volatile Organic Compounds

Little is known about the fate of the SVOC, 2-butoxyethanol (2-BE) (an antifoaming and anti-corrosion agent used in slick-water) (Volz et al., 2011) or its potential impact on surface waters, drinking water resources, or drinking water systems. This compound is very soluble in water and is subject to biodegradation, with a half-life estimation of 1-4 weeks in the environment (Wess et al., 1998). The EPA has not classified 2-BE (or other glycol ethers) for carcinogenicity. 2-BE was detected in the discharge of a Pennsylvania CWT at concentrations of 59 mg/L (Volz et al., 2011). Ferrar et al. (2013) detected 2-BE in the effluents from a CWT in western Pennsylvania at average concentrations of 34 – 45 mg/L; the latter value was measured when the CWT was receiving only conventional oil and gas wastewater. Data are lacking on 2-BE concentrations in surface waters that receive treated effluents from hydraulic fracturing wastewater treatment systems.

Polycyclic aromatic hydrocarbons are another common group of semi-volatile organic compounds (SVOCs) in oil and gas wastewater. They have been detected in soils 164 feet (50 m) downgradient of an unlined pit in New Mexico (Sumi, 2004; Eiceman, 1986). PAHs were also found in birds in wetlands fed by oil and gas wastewater discharges in Wyoming (Ramirez, 2002).

8.6.6. Oil and Grease

Oil and gas wastewater often contains oil and grease from the formation or from oil-based drilling fluids. Typically, oil and grease are separated from the wastewater before discharge either by a heat treatment or by allowing gravity separation followed by skimming. If these processes are inefficient, oil and grease may be integrated with the discharge to surface waters. For example, in some cases, oil and grease are allowed to separate in pits, and water is then withdrawn from the lower part of the pit with a standpipe. If the oil layer is allowed to drop to the level of the standpipe or if the water is agitated, oil and grease may be discharged along with the water. Oil and grease are also often dispersed in wastewater in the form of small droplets that are 4 to 6 microns in diameter. These droplets can be difficult to remove using typical oil/water separators (Veil et al., 2004). In a study by the U.S. Fish and Wildlife Service regarding permitted oil and gas discharges between 1996 and 1999 from Wyoming oil fields, 15% of the 62 discharges to Wyoming wetlands reviewed showed visible oil sheens in the receiving water and 10 of the sites sampled exceeded discharge limits of 10 mg/L of oil and grease (Ramirez, 2002).

8.7. Synthesis

Hydraulic fracturing operations produce fluids during the flowback and production phases (collectively called wastewater) of a production well, along with liquid and solid treatment residuals from treatment processes. A variety of management strategies may be considered, with cost frequently a driving factor. Available information suggests that Class IID wells regulated under the Underground Injection Control (UIC) Program are the most frequently used wastewater management practice, but reuse, discharge after treatment, and various other uses are also employed.

8.7.1. Summary of Findings

Hundreds of billions of gallons of wastewater are generated annually in the United States by the oil and gas industry, although national level estimates are difficult to reliably obtain. It is also difficult to produce a nationwide estimate of the amount of wastewater that is attributable specifically to hydraulic fracturing because some states do not specifically identify wastewater from hydraulic fracturing operations in their available wastewater data.

The total amount of wastewater produced in an area corresponds generally to oil and gas production and, therefore, may increase if hydrocarbon production increases in a region. Geographically, a large portion of oil and gas wastewater in the United States is reported to be generated in the western part of the country, including contributions from both conventional and unconventional resources. For some states, estimates of hydraulic fracturing wastewater volumes can be made using publicly available production or waste data. Annual estimates compiled in this way range from hundreds of millions to billions of gallons of wastewater generated per state per year. Direct comparisons among these state data are problematic, however, because of a great deal of variability in state data collection, including differences in the years for which data are available, and challenges in definitively identifying wells that have been hydraulically fractured (to distinguish hydraulic fracturing wastewater from that generated from wells that are not hydraulically fractured). Within a given state, however, estimated volumes in areas where hydraulic fracturing is practiced extensively have generally increased over the last several years, along with numbers of wells contributing to total wastewater volumes. For example, the data made available by PA DEP illustrate that the total volume of wastewater generated correlates generally with a significant increase in volume of hydrocarbon production and with the number of production wells. As hydraulic fracturing activities increase and the number of wells increases, the amount of hydraulic fracturing wastewater generated is likely to increase.

8.7.1.1. Wastewater Management Practices

Hydraulic fracturing wastewater is managed in a variety of ways, including disposal through Class IID wells; minimal treatment and reuse in subsequent fracturing operations; more complete treatment followed by discharge, disposal, or reuse; evaporation; and other uses such as irrigation (when the wastewater quality is adequate). Unauthorized discharges of hydraulic fracturing wastewaters have been documented; such discharges could potentially impact drinking water resources, but estimates of the frequency of occurrence cannot be developed with the available data.

As of 2015, available information suggests that wastewater management practices involve extensive use of Class II wells to manage wastewater from most of the major unconventional plays in the United States, with the notable exception of the Marcellus Shale region in Pennsylvania. More than 98% of wastewater in the oil and gas industry is estimated to be injected into Class II wells annually (including wells for enhanced oil recovery and disposal) (Clark and Veil, 2009). Based on data compiled from 2012 and 2014, there are about 25,000 Class IID wells in the United States (U.S. EPA, 2015q). In particular, large numbers of active injection wells are found in Texas (7,876 or

29%), Kansas (5,516 or 20%), Oklahoma (4,622 or 17%), Louisiana (2,448 or 9%), and Illinois (1,054 or 4%).

Use of Class IID wells is likely driven by the availability of Class IID wells within reasonable transportation distance and the cost of transporting (and injecting) the wastewaters. In the oil and gas industry, Class IID wells have generally been the most economically favorable wastewater management practice (U.S. GAO, 2012). In Pennsylvania, there are only nine Class IID wells as of February 2015, and a significant growth of gas production using hydraulic fracturing in the Marcellus is generating increasing amounts of wastewater. Treatment and reuse are becoming increasingly popular in the Marcellus Shale region and are in more widespread use in comparison to other oil and gas producing parts of the country.

Reuse of hydraulic fracturing wastewater to formulate fracturing fluid in subsequent hydraulic fracturing jobs varies considerably on a national level, and reliable estimates are not available for all areas. As of 2014–2015, the greatest amount of reuse occurs in Pennsylvania, where the scarcity of Class IID wells to receive Marcellus wastewater drives this practice. Recent estimates of wastewater reuse in Pennsylvania range as high as 90% or more. Waste disposal data from the PA DEP (2015a) indicate that much of the reuse happens on-site. Operators also report some reuse of wastewater in other regions such as the Haynesville Shale, the Fayetteville Shale, the Barnett Shale, and the Eagle Ford Shale, although at much lower volume percentages (about 5 – 20%) compared with practices in the Marcellus Shale region. Increased reuse and recycling of hydraulic fracturing wastewaters has the added benefit of providing an additional water supply for hydraulic fracturing fluid formulation in areas where water scarcity is a concern. If, however, hydraulic fracturing activity slows, demand for wastewater for reuse will also decrease, and other forms of wastewater management will be needed.

The decision to reuse/recycle depends upon several factors, including the volume and rate of production of the wastewater and whether these are suitable for water needs for ongoing fracturing activities in the area. The composition of the water, in particular the TSS and TDS content, and whether the water quality can be accommodated in the fracturing practices in an area can also influence reuse, including decisions about what type of pretreatment or treatment may be needed to make reuse or recycling feasible.

Treatment facilities (either centralized waste treatment facilities (CWTs) or systems designed for on-site use) can be permitted to treat oil and gas wastewaters. Treatment can be followed by discharge to a surface water body or to a POTW, or the treated effluent may be used for reuse. Most CWTs treating hydraulic fracturing wastewater are located in Pennsylvania (39 facilities), and a number of CWTs (11) are located in Ohio. More are under construction or pending approval. Most are "zero-discharge" and do not have the treatment capacity to reduce TDS; their effluent is reused for hydraulic fracturing. Specialized on-site, mobile, or semi-mobile treatment facilities can be used by operators to handle wastewater without the expense of long-distance transportation and can be customized to produce an effluent that meets the water quality needs of the intended disposal or reuse plans.

Treatment of hydraulic fracturing wastewaters by publicly owned treatment works (POTWs) was previously practiced in Pennsylvania. POTWs are not designed for the high TDS content of Marcellus wastewaters, and stricter discharge limits for TDS in Pennsylvania, as well as a positive response to a request from Pennsylvania DEP that operators stop sending Marcellus wastewater to POTWs and some CWTs, led to the practice being discontinued in 2011. (Some POTWs in Pennsylvania still accept oil and gas wastewaters from conventional operations, including conventional wells that have undergone hydraulic fracturing.)

Management plans will necessarily need to change with time as hydraulic fracturing activities in a region change. The volumes of wastewater also change during the life of a well. The chemical composition of the wastewater changes during the transition from the flowback period and into the production phase. In addition, the demand for reused water to support ongoing fracturing activities will change. Taken in aggregate, these factors may influence costs and choices associated with hydraulic fracturing wastewater management, especially if Class IID wells are limited in a particular area for any reason.

8.7.1.2. Treatment and Discharge

One of the most frequently cited concerns regarding hydraulic fracturing wastewater, especially from shale plays and tight sand plays, is the high TDS content, which poses challenges for treatment, discharge, and reuse. Conventional treatment processes such as sedimentation, filtration methods, flotation, chemical precipitation and ion exchange can remove constituents such as oil and grease, major cations, metals, and TSS. Because these processes do not remove monovalent ions (e.g., chloride, bromide, sodium), reducing TDS in these high-salinity wastewaters requires more advanced processes such as reverse osmosis (RO), electrodialysis, and distillation methods. Distillation methods appear to be the approach of choice for newer CWT facilities that are designed to lower TDS. RO, while highly effective, does have limits to TDS concentrations (less than approximately 40,000 mg/L) that it can treat (Shaffer et al., 2013; Younos and Tulou, 2005).

Hydraulic fracturing wastewater discharged from treatment facilities without advanced TDS removal processes has been shown to cause elevated TDS, bromide, and chloride levels in receiving waters in Pennsylvania. Existing literature indicates that bromide and chloride are important wastewater constituents with regard to potential burdens on downstream drinking water treatment facilities. Bromide in particular is of concern due to the formation of disinfection by-products (DBPs) during disinfection. Some types of DBPs are regulated under SDWA's Stage 1 and Stage 2 DBP Rules, but a subset of DBPs, including a number of chlorinated, brominated, nitrogenous, and iodinated DBPs, are not regulated. Brominated DBPs (and iodinated DBPs) are more toxic than other species of DBPs. Modeling suggests that very small percentages of hydraulic fracturing wastewater in a river used as a source for drinking water treatment plants may cause a notable increase in DBP formation.

Radionuclides (in particular radium-226 and radium-228) in some hydraulic fracturing wastewaters pose concerns for the quality of discharges if they are not adequately treated. Possible elevated radionuclide content in treatment residuals is also a consideration. In Marcellus Shale gas production wastewater, radium-226, radium-228, gross alpha, and gross beta are most cited as the radioactive constituents of concern, and concentrations can range up to thousands of pCi/L. Fewer data exist on uranium content in wastewaters, and data are also limited on radionuclide concentrations in wastewaters from other unconventional plays. A confounding issue in evaluating radium concentrations is underestimation when using traditional wet chemical methods with high-TDS waters. A variety of treatment processes can be used for removal of radium, ranging from conventional methods such as chemical precipitation and filtration to more advanced and costly techniques, such as reverse osmosis or distillation (including mechanical vapor recompression). Whether the effluent from such treatment contains elevated radium, however, will depend upon influent concentrations as well as treatment removal efficiency.

Other potential effects on drinking water resources may result from discharges or spills of hydraulic fracturing wastewaters containing elevated concentrations of barium and other metals. Again, the management strategy and treatment choices will affect the likelihood of such impacts.

8.7.2. Factors Affecting the Frequency or Severity of Impacts

On a regional scale, potential effects on drinking water resources from hydraulic fracturing wastewater will depend upon the mix of wastewater management strategies used, and potential impacts may change through time if the quantity of hydraulic fracturing wastewater changes and strategies to manage the wastewater change. For example, if use of Class IID wells becomes restricted in parts of the country where they are currently commonly used, the emphasis may shift, at least locally, from use of Class IID wells and towards the use of treatment and either discharge or reuse. Although reuse delays the discharge of wastewater by directing it to ongoing fracturing activities, reuse may ultimately concentrate constituents such as radionuclides (depending upon the ratio of recycled to new water). If a stream of wastewater or portion of wastewater has been used for more than one hydraulic fracturing event and is eventually intended for disposal, the method of disposal will need to be appropriate for the quality of the wastewater.

Potential effects on drinking water resources from hydraulic fracturing wastewaters that undergo treatment depend upon the quality and quantity of discharges to receiving waters (discharge could occur directly after treatment at a CWT or indirectly after discharge to a POTW). Hydraulic fracturing wastewater management can consider appropriate levels of treatment and blending so that the resulting TDS content in a receiving water will not result in formation of DBPs during subsequent drinking water treatment and will not impair biological treatment processes.

The volumes of discharges relative to the receiving water body size are important local factors to consider in evaluating whether elevated concentrations can be anticipated at downstream drinking water intakes. Small drinking water systems drawing water from smaller streams in affected areas would likely face greater challenges in dealing with high bromide and chloride levels in source waters. Furthermore, other potential impacts on surface water and shallow ground water may exist due to spills of either untreated wastewater or effluent from zero-discharge CWTs, and there will be site-specific factors such as distance to a water body or depth to the water table to consider (see Chapter 5).

Results from existing literature and recent PA DEP data suggest that cumulative impacts from radionuclides may occur in sediments at or near discharge points from facilities that treat and discharge oil and gas wastewater (or have done so in the past). There may be consequences for downstream drinking water systems if the sediments are disturbed or entrained due to dredging or flood events. Similarly, some organic constituents may not be removed during treatment, and potential effects on receiving waters and sediments will depend upon the properties of the specific constituents, their concentrations, and the treatment used.

The possibility of radionuclides affecting receiving waters and sediments will depend upon the technologically enhanced naturally occurring radioactive material (TENORM) content of the wastewater and the treatment processes used. Although radionuclide contamination at drinking water intakes due to treated hydraulic fracturing fluid has not been detected, a recent PA DEP study (PA DEP, 2015b) has revealed radium in effluents from both CWTs handling oil and gas wastewater and POTWs receiving effluent from such facilities. The concentrations in the CWT effluents were considerably higher than in the POTW effluents. The site selection criteria for this study included some Pennsylvania wastewater facilities whose influents include wastewater from unconventional operations or where radioactivity was measured in the influent, sludges, or effluents (CWTs may also receive conventional wastewater). In regions where unconventional plays are known to be enriched in radionuclides, analysis of TENORMs in untreated hydraulic fracturing wastewaters, selection of appropriate treatment processes, and monitoring of TENORMs in treatment effluent and receiving waters could help address potential impacts on drinking water resources. Gross alpha and gross beta measurements or gamma spectroscopic analyses could be used as initial screening methods for radionuclides. Enrichment of TENORMs in waste products from treatment processes also requires appropriate management to reduce potential impacts.

Other management strategies such as irrigation, road spreading, and evaporation are less frequently employed for hydraulic fracturing wastewaters. Irrigation or land application may have potential effects on surface waters depending upon the constituents in the wastewater (e.g., salts and radionuclides), the distance from the site of application to a receiving water, and whether stormwater management measures exist that mitigate runoff. Distance to the water table, precipitation, and the hydrogeologic properties of the soil and sediment will influence whether migration of these constituents results in contamination of shallow ground water.

8.7.3. Uncertainties

A full understanding of the practices being used for management of hydraulic fracturing wastewaters is limited by a lack of available data in a number of areas. It is difficult to assemble a complete, national- or regional-level picture of wastewater generation and management practices because the tracking and availability of data vary from state to state. Although some states provide well-organized and relatively thorough data, not all states make such information available, and it can be difficult to identify wastewater volumes specifically associated with hydraulic fracturing activity (as compared to all oil and gas production activities). Such data would be needed to place hydraulic fracturing wastewater in the broader context of all oil and gas wastewaters. Data are also generally difficult to locate for production volumes, chemical composition, masses, and management and disposal strategies for residuals.

Among management practices, up-to-date information on the volumes of hydraulic fracturing wastewaters disposed of via underground injection in different states are not uniformly available. Without this information, it is difficult to assess whether disposal well capacity will become an issue in areas where hydraulic fracturing activity is expected to increase.

Assessment of the potential effects of hydraulic fracturing on drinking water resources is also limited by relatively few data on effluent quality from CWTs that receive oil and gas wastewaters (including those associated with hydraulic fracturing) and POTWs that receive CWT effluents. If a CWT can discharge to surface water (e.g., the CWT has a NPDES permit), some monitoring data may be available that will provide information on effluent quality, but the list of monitored constituents may be limited.

In evaluating the treatment effectiveness of full scale facilities, relatively few data exist on the quality of both influents and effluents from treatment facilities, although some manufacturers of patented CWT systems publicize information on treatment effectiveness. A better understanding of the pollutant removal capabilities of facilities would be helped by influent and effluent sampling, timed so that effluent samples are representative of influent samples to the degree possible. There are limited analyses of influent and effluent samples for a wide range of constituents associated with hydraulic fracturing fluids and wastewaters (e.g., major cations and anions, radionuclides, metals, VOCs, SVOCs, diesel range organics (DROs), and total petroleum hydrocarbons (TPH)). Analyses are needed in which the methods are appropriate for the TDS content of the sample. Radium in particular needs to be analyzed using a method suitable for high-salt samples, otherwise concentrations may be underestimated. Continued work towards ensuring that analytical methods exist for the highly complex matrixes often encountered with oil and gas wastewater would provide better certainty in the results of chemical analyses.

Monitoring of surface waters, even screening with a simple TDS proxy such as conductivity, would be needed to help assess how often hydraulic fracturing activities (including spills or discharges of wastewater) affect receiving waters; such data are lacking except for some studies in the Marcellus Shale region. Existing data are also limited regarding legacy effects, such as accumulation of contaminants in sediments at discharge points, soil accumulation due to application of de-icing brines or salts from wastewater treatment, and handling of waste water treatment residuals.

8.7.4. Conclusions

Oil and gas operations in the United States generate billions of gallons of wastewater daily; this includes wastewater associated with hydraulic fracturing activities, although what portion of this oil and gas wastewater is attributable to hydraulic fracturing operations is difficult to estimate due to lack of consistent data regarding wastewater volumes. Available information indicates that the majority of this water is injected into Class IID wells regulated under the Underground Injection Control (UIC) program, although in some areas of the country, wastewater is reused (either with our without treatment) for new hydraulic fracturing jobs. In the Marcellus Shale region in

Pennsylvania, the majority of wastewater is currently reused. Wastewater may also be treated in a CWT and discharged to a surface water body or to a POTW, or in certain settings, used for various other uses (e.g., irrigation) if water quality allows. Impacts on drinking water resources may result from inadequate treatment prior to discharge or spills. Particular constituents of concern in wastewater from hydraulic fracturing, especially in the Marcellus Shale region, include bromide and radionuclides. There is limited information regarding the influents and effluents from facilities that treat wastewater from hydraulic fracturing operations.

Text Box 8-2. Research Questions Revisited.

What are the common treatment and disposal methods for hydraulic fracturing wastewater, and where are these methods practiced?

- The majority of hydraulic fracturing wastewater in the United States is disposed of via underground injection wells. As of 2014-2015, most states where hydraulic fracturing occurs have an adequate number of Class IID injection wells regulated under the Underground Injection Control (UIC) program. The Marcellus Shale region, especially the northeastern region, is an exception. Wastewater treatment for reuse is increasing in the Marcellus shale region and may continue to increase in western shale plays as the practice becomes encouraged and economically favorable.

How effective are conventional POTWs and commercial treatment systems in removing organic and inorganic contaminants of concern in hydraulic fracturing wastewater?

- Publicly owned treatment works (POTWs) using basic treatment processes cannot effectively reduce elevated total dissolved solids (TDS) concentrations in hydraulic fracturing wastewater. Centralized waste treatment facilities (CWTs) with advanced treatment processes can remove TDS constituents with removal efficiencies ranging from 97% to over 99% as demonstrated at facilities that use treatment processes such as mechanical vapor recompression, distillation, and reverse osmosis (see Table 8-6). Advanced treatment processes have been shown to remove certain contaminants found in hydraulic fracturing wastewater (see Table 8-6). Indirect discharge, where wastewater is pretreated by a CWT and sent to a POTW, may be an effective option for hydraulic fracturing wastewater treatment (with restrictions on contaminant concentrations in the pretreated wastewater that is sent to a POTW). This option would require careful planning to ensure that the pretreated wastewater blended with POTW influent is of appropriate quality and quantity to prevent deleterious effects on biological processes in the POTW or the pass-through of contaminants.

- Facilities that treat wastewater for reuse and employ only basic treatment are unable to remove all contaminants in hydraulic fracturing wastewater. Depending on the water quality requirements for a particular site, these lower quality treated waters may be of adequate quality for reuse on subsequent hydraulic fracturing operations (and will be less costly). Some organic compounds (BTEX, some alcohols, 2-butoxyethanol) may not be removed by the processes employed in CWTs if they don’t include specific processes that target these compounds (e.g., distillation, advanced oxidation, adsorption).

What are the potential impacts on drinking water treatment facilities from surface water disposal of treated hydraulic fracturing wastewater?

- Inadequate bromide and iodide removal from treated hydraulic fracturing wastewater has the greatest potential to affect surface water quality and place a burden on downstream drinking water treatment facilities that use chlorine-based disinfection due to the formation of DBPs. Radionuclides, metals, and trace organic compounds in effluents from CWTs may also be of concern if present in treated wastewater or if they accumulate in sediments downstream of discharge points. These constituents have reached drinking water resources via some discharges, although sampling data for effluents and receiving waters are limited. As of 2014-2015, there is no evidence that these contaminants have affected drinking water facilities, but data are lacking for concentrations of these constituents at drinking water intakes in regions with hydraulic fracturing.

8.8. References for Chapter 8

Alanco. (2012). New subsidiary Alanco Energy Services, Inc. to provide produced water disposal services to natural gas industry. Alanco. http://www.alanco.com/news_040912.asp

Alzahrani, S; Mohammad, AW; Hilal, N; Abdullah, P; Jaafar, O. (2013). Comparative study of NF and RO membranes in the treatment of produced water-Part I: Assessing water quality. Desalination 315: 18-26. http://dx.doi.org/10.1016/j.desal.2012.12.004

API (American Petroleum Institute). (2000). Overview of exploration and production waste volumes and waste management practices in the United States. http://www.api.org/environment-health-and-safety/environmental-performance/~/media/Files/EHS/Environmental_Performance/ICF-Waste-Survey-of-EandP-Wastes-2000.ashx

Argonne National Laboratory. (2014). Water use and management in the Bakken shale oil play. (DOE Award No.: FWP 49462). Pittsburgh, PA: National Energy Technology Laboratory. http://www.ipd.anl.gov/anlpubs/2014/05/104645.pdf

Bacher, D. (2013). Oil company fined $60,000 for illegally discharging fracking fluid. Available online at https://www.indybay.org/newsitems/2013/11/17/18746493.php?show_comments=1 (accessed March 6, 2015).

Bair, ES; Digel, RK. (1990). Subsurface transport of inorganic and organic solutes from experimental road spreading of oilfield brine. Ground Water Monitoring and Remediation 10: 94-105.

Banasiak, LJ; Schäfer, AI. (2009). Removal of boron, fluoride and nitrate by electrodialysis in the presence of organic matter. J Memb Sci 334: 101-109. http://dx.doi.org/10.1016/j.memsci.2009.02.020

Barbot, E; Vidic, NS; Gregory, KB; Vidic, RD. (2013). Spatial and temporal correlation of water quality parameters of produced waters from Devonian-age shale following hydraulic fracturing. Environ Sci Technol 47: 2562-2569.

Benko, KL; Drewes, JE. (2008). Produced water in the Western United States: Geographical distribution, occurrence, and composition. Environ Eng Sci 25: 239-246.

Blauch, ME; Myers, RR; Moore, TR; Lipinski, BA. (2009). Marcellus shale post-frac flowback waters - where is all the salt coming from and what are the implications? In Proceedings of the SPE Eastern Regional Meeting. Richardson, TX: Society of Petroleum Engineers.

Boschee, P. (2012). Handling produced water from hydraulic fracturing. Oil and Gas Facilities 1: 23-26.

Boschee, P. (2014). Produced and flowback water recycling and reuse: Economics, limitations, and technology. Oil and Gas Facilities 3: 16-22.

Bowen, ZH; Oelsner, GP; Cade, BS; Gallegos, TJ; Farag, AM; Mott, DN; Potter, CJ; Cinotto, PJ; Clark, ML; Kappel, WM; Kresse, TM; Melcher, CP; Paschke, SS; Susong, DD; Varela, BA. (2015). Assessment of surface water chloride and conductivity trends in areas of unconventional oil and gas development-Why existing national data sets cannot tell us what we would like to know. Water Resour Res 51: 704-715. http://dx.doi.org/10.1002/2014WR016382

Boysen, DB; Boysen, JA; Boysen, JE. (2002). Creative Strategies for Produced Water Disposal in the Rocky Mountain Region. Paper presented at 9th Annual International Petroleum Environmental Conference, October 2002, Albuquerque, NM.

Boysen, JE; Harju, JA; Shaw, B; Fosdick, M; Grisanti, A; Sorensen, JA. (1999). The current status of commercial deployment of the freeze thaw evaporation treatment of produced water. Richardson, TX: Society of Petroleum Engineers. http://dx.doi.org/10.2118/52700-MS

Bruff, M; Jikich, SA. (2011). Field demonstration of an integrated water treatment technology solution in Marcellus shale. Paper presented at SPE Eastern Regional Meeting, August 17-19, 2011, Columbus, OH.

California Department of Conservation. (2015). Monthly production and injection databases. statewide production and injection data [Database]. Sacramento, CA: California Department of Conservation, Division of Oil, Gas & Geothermal Resources. Retrieved from http://www.conservation.ca.gov/dog/prod_injection_db/Pages/Index.aspx

Camacho, LM, ar; Dumee, L; Zhang, J; Li, J; Duke, M; Gomez, J; Gray, S. (2013). Advances in membrane distillation for water desalination and purification applications. Water 5: 94-196. http://dx.doi.org/10.3390/w5010094

Chapman, EC; Capo, RC; Stewart, BW; Kirby, CS; Hammack, RW; Schroeder, KT; Edenborn, HM. (2012). Geochemical and strontium isotope characterization of produced waters from Marcellus Shale natural gas extraction. Environ Sci Technol 46: 3545-3553.

Chester Engineers. (2012). The Pittsburgh water and sewer authority 40 year plan. (PWSA Project No. R-D1.10015-11). Pittsburgh, PA: The Pittsburgh Water and Sewer Authority. http://apps.pittsburghpa.gov/pwsa/PWSA_40-year_Plan.pdf

Chiado, ED. (2014). The impact of shale gas/oil waste on MSW landfill composition and operations. In CL Meehan; JM VanBriesen; F Vahedifard; X Yu; C Quiroga (Eds.), Shale energy engineering 2014 technical challenges, environmental issues, and public policy (pp. 412-420). Reston, VA: American Society of Civil Engineers. http://dx.doi.org/10.1061/9780784413654.044

Clark, CE; Horner, RM; Harto, CB. (2013). Life Cycle Water Consumption for Shale Gas and Conventional Natural Gas. Environ Sci Technol 47: 11829-11836. http://dx.doi.org/10.1021/es4013855

Clark, CE; Veil, JA. (2009). Produced water volumes and management practices in the United States (pp. 64). (ANL/EVS/R-09/1). Argonne, IL: Argonne National Laboratory. http://www.circleofblue.org/waternews/wp-content/uploads/2010/09/ANL_EVS__R09_produced_water_volume_report_2437.pdf

COGCC (Colorado Oil and Gas Conservation Commission). (2015). COGIS - all production reports to date. Denver, CO. Retrieved from http://cogcc.state.co.us/

Colorado Division of Water Resources; Colorado Water Conservation Board; Colorado Oil and Gas Conservation Commission. (2014). Water sources and demand for the hydraulic fracturing of oil and gas wells in Colorado from 2010 through 2015 [Fact Sheet]. http://cewc.colostate.edu/2012/02/water-sources-and-demand-for-the-hydraulic-fracturing-of-oil-and-gas-wells-in-colorado-from-2010-through-2015/

Countess, S; Boardman, G; Hammack, R; Hakala, A; Sharma, S; Parks, J. (2014). Evaluating leachability of residual solids from hydraulic fracturing in the Marcellus shale. In Shale energy engineering 2014: Technical challenges, environmental issues, and public policy. Reston, VA: American Society of Civil Engineers. http://dx.doi.org/10.1061/9780784413654.012

Cusick, M. (2013). EPA fines western PA treatment plants for Marcellus wastewater violations. Available online at http://stateimpact.npr.org/pennsylvania/2013/05/24/epa-fines-western-pa-treatment-plants-for-marcellus-wastewater-violations/ (accessed March 6, 2015).

Dahm, K; Chapman, M. (2014). Produced water treatment primer: Case studies of treatment applications. (S&T Research Project #1617). Denver CO: U.S. Department of the Interior. http://www.usbr.gov/research/projects/download_product.cfm?id=1214.

DOE (U.S. Department of Energy). (2002). Handbook on best management practices and mitigation strategies for coalbed methane in the Montana portion of the Powder River Basin. Tulsa, OK: U.S. Department of Energy, National Energy Technology Laboratory. http://bogc.dnrc.mt.gov/PDF/BMPHandbookFinal.pdf

DOE (U.S. Department of Energy). (2003). Handbook on coalbed methane produced water: Management and beneficial use alternatives. Tulsa, OK: ALL Consulting. http://www.all-llc.com/publicdownloads/CBM_BU_Screen.pdf

DOE (U.S. Department of Energy). (2004). A white paper describing produced water from production of crude oil, natural gas, and coal bed methane. Lemont, IL: Argonne National Laboratory. http://seca.doe.gov/technologies/oil-gas/publications/oil_pubs/prodwaterpaper.pdf

DOE (U.S. Department of Energy). (2006). A guide to practical management of produced water from onshore oil and gas operations in the United States. Washington, DC: U.S. Department of Energy, National Petroleum Technology Office. http://fracfocus.org/sites/default/files/publications/a_guide_to_practical_management_of_produced_water_from_onshore_oil_and_gas_operations_in_the_united_states.pdf

Drewes, J; Cath, T; Debroux, J; Veil, J. (2009). An integrated framework for treatment and management of produced water - Technical assessment of produced water treatment technologies (1st ed.). (RPSEA Project 07122-12). Golden, CO: Colorado School of Mines. http://aqwatec.mines.edu/research/projects/Tech_Assessment_PW_Treatment_Tech.pdf

Duraisamy, RT; Beni, AH; Henni, A. (2013). State of the art treatment of produced water. In W Elshorbagy; RK Chowdhury (Eds.), Water treatment (pp. 199-222). Rijeka, Croatia: InTech. http://dx.doi.org/10.5772/53478

Easton, J. (2014). Optimizing fracking wastewater management. Pollution Engineering January 13.

EIA (Energy Information Administration). (2014c). Natural gas. U.S. Energy Information Administration: Independent statistics and analysis. Available online at http://www.eia.gov/naturalgas/

Eiceman, GA. (1986). Hazardous organic wastes from natural gas production, processing and distribution: Environmental fates. (WRRI report, no. 227). New Mexico: Water Resources Research Institute. http://wrri.nmsu.edu/publish/techrpt/abstracts/abs227.html

Engle, MA; Bern, CR; Healy, RW; Sams, JI; Zupancic, JW; Schroeder, KT. (2011). Tracking solutes and water from subsurface drip irrigation application of coalbed methaneproduced waters, Powder River Basin, Wyoming. Environmental Geosciences 18: 169-187.

Entrekin, S; Evans-White, M; Johnson, B; Hagenbuch, E. (2011). Rapid expansion of natural gas development poses a threat to surface waters. Front Ecol Environ 9: 503-511. http://dx.doi.org/10.1890/110053

EPA (Environmental Protection Agency). (2000). Development document for effluent limitations guidelines and standards for the centralized waste treatment industry. (821R00020). Washington, DC: U.S. Environmental Protection Agency.

Ertel, D; McManus, K; Bogdan, J. (2013). Marcellus wastewater treatment: Case study. In Summary of the technical workshop on wastewater treatment and related modeling (pp. A56-A66). Williamsport, PA: Eureka Resources, LLC. http://www2.epa.gov/hfstudy/summary-technical-workshop-wastewater-treatment-and-related-modeling

Fakhru'l-Razi, A; Pendashteh, A; Abdullah, LC; Biak, DR; Madaeni, SS; Abidin, ZZ. (2009). Review of technologies for oil and gas produced water treatment [Review]. J Hazard Mater 170: 530-551.

Ferrar, KJ; Michanowicz, DR; Christen, CL; Mulcahy, N; Malone, SL; Sharma, RK. (2013). Assessment of effluent contaminants from three facilities discharging Marcellus Shale wastewater to surface waters in Pennsylvania. Environ Sci Technol 47: 3472-3481.

Geiver, L. (2013). Frac water treatment yields positive results for Houston Co. Retrieved from http://www.thebakken.com/articles/20/frac-water-treatment-yields-positive-results-for-houston-co

Gomes, J; Cocke, D; Das, K; Guttula, M; Tran, D; Beckman; J. (2009). Treatment of produced water by electrocoagulation. Shiner, TX: KASELCO, LLC. http://www.kaselco.com/index.php/library/industry-white-papers

Gonneea, ME; Morris, PJ; Dulaiova, H; Charette, MA. (2008). New perspectives on radium behavior within a subterranean estuary. Mar Chem 109: 250-267. http://dx.doi.org/10.1016/j.marchem.2007.12.002

Greenhunter (Greenhunter Resources). (2014). Oillfield water management solutions. Available online at http://www.greenhunterenergy.com/operations/owms.htm

Gregory, KB; Vidic, RD; Dzombak, DA. (2011). Water management challenges associated with the production of shale gas by hydraulic fracturing. Elements 7: 181-186.

GTI (Gas Technology Institute). (2012). Barnett and Appalachian shale water management and resuse technologies. (Report no. 08122-05.FINAL.1). Sugar Land, TX: Research Partnership to Secure Energy for America, RPSEA. https://www.netl.doe.gov/file%20library/research/oil-gas/Natural%20Gas/shale%20gas/08122-05-final-report.pdf

Guerra, K; Dahm, K; Dundorf, S. (2011). Oil and gas produced water management and beneficial use in the western United States. (Science and Technology Program Report No. 157). Denver, CO: U.S. Department of the Interior Bureau of Reclamation.

Halliburton. (2014). Hydraulic fracturing 101. Available online at http://www.halliburton.com/public/projects/pubsdata/hydraulic_fracturing/fracturing_101.html

Hamieh, BM; Beckman, JR. (2006). Seawater desalination using Dewvaporation technique: theoretical development and design evolution. Desalination 195: 1-13. http://dx.doi.org/10.1016/j.desal.2005.09.034

Hammer, R; VanBriesen, J. (2012). In frackings wake: New rules are needed to protect our health and environment from contaminated wastewater. New York, NY: Natural Resources Defense Council. http://www.nrdc.org/energy/files/fracking-wastewater-fullreport.pdf

Hansen, E; Mulvaney, D; Betcher, M. (2013). Water resource reporting and water footprint from Marcellus Shale development in West Virginia and Pennsylvania. Durango, CO: Earthworks Oil & Gas Accountability Project. http://www.downstreamstrategies.com/documents/reports_publication/marcellus_wv_pa.pdf

Hayes, T. (2009). Sampling and analysis of water streams associated with the development of Marcellus shale gas. Des Plaines, IL: Marcellus Shale Coalition. http://eidmarcellus.org/wp-content/uploads/2012/11/MSCommission-Report.pdf

Hayes, T; Severin, BF. (2012b). Evaluation of the aqua-pure mechanical vapor recompression system in the treatment of shale gas flowback water - Barnett and Appalachian shale water management and reuse technologies. (08122-05.11). Hayes, T; Severin, BF. http://barnettshalewater.org/documents/08122-05.11-EvaluationofMVR-3-12-2012.pdf

Hayes, TD; Halldorson, B; Horner, P; Ewing, J; Werline, JR; Severin, BF. (2014). Mechanical vapor recompression for the treatment of shale-gas flowback water. Oil and Gas Facilities 3: 54-62.

Hladik, ML; Focazio, MJ; Engle, M. (2014). Discharges of produced waters from oil and gas extraction via wastewater treatment plants are sources of disinfection by-products to receiving streams. Sci Total Environ 466. http://dx.doi.org/10.1016/j.scitotenv.2013.08.008

Hulme, D. (2005). CBM co-produced water management, disposal, treatment and use. Hulme, D. http://www.uwyo.edu/haub/ruckelshaus-institute/_files/docs/publications/2005-cbm-water-management-hulme.pdf

IAEA (International Atomic Energy Agency). (2014). The environmental behaviour of radium: revised edition. Vienna, Austria. http://www-pub.iaea.org/MTCD/Publications/PDF/trs476_web.pdf

Igunnu, ET; Chen, GZ. (2014). Produced water treatment technologies. International Journal of Low-Carbon Technologies 9: 157-177. http://dx.doi.org/10.1093/ijlct/cts049

IHS (Global Insight). (2013). Americas new energy future: The unconventional oil and gas revolution and the US economy. Douglas County, Colorado. http://www.energyxxi.org/sites/default/files/pdf/Americas_New_Energy_Future_Phase3.pdf

Kappel, WM; Williams, JH; Szabo, Z. (2013). Water resources and shale gas/oil production in the Appalachian basin critical issues and evolving developments. (Open-File Report 20131137). Troy, NY: U.S. Geological Survey. http://pubs.usgs.gov/of/2013/1137/pdf/ofr2013-1137.pdf

Kargbo, DM; Wilhelm, RG; Campbell, DJ. (2010). Natural gas plays in the Marcellus Shale: Challenges and potential opportunities. Environ Sci Technol 44: 5679-5684. http://dx.doi.org/10.1021/es903811p

Kaushal, SS; Groffman, PM; Likens, GE; Belt, KT; Stack, WP; Kelly, VR; Band, LE; Fisher, GT. (2005). Increased salinization of fresh water in the northeastern United States. PNAS 102: 13517-13520. http://dx.doi.org/10.1073/pnas.0506414102

Kelly, WR. (2008). Long-term trends in chloride concentrations in shallow aquifers near Chicago. Ground Water 46: 772-781. http://dx.doi.org/10.1111/j.1745-6584.2008.00466.x

Kennedy/Jenks Consultants. (2002). Evaluation of technical and economic feasibility of treating oilfield produced water to create a new water resource. http://www.gwpc.org/sites/default/files/event-sessions/Roger_Funston_PWC2002_0.pdf

Kharaka, YK; Kakouros, E; Abbott, MM. (2002). Environmental impacts of petroleum production: 1- The fate of inorganic and organic chemicals in produced water from the Osage-Skiatook Petroleum Environmental Research B site, Osage County, OK. 9th International Petroleum Environmental Conference, October 22-25, 2002, Albuquerque, NM.

Krasner, SW. (2009). The formation and control of emerging disinfection by-products of health concern [Review]. Philos Transact A Math Phys Eng Sci 367: 4077-4095. http://dx.doi.org/10.1098/rsta.2009.010

Kuwayama, Y; Olmstead, S; Krupnick, A. (2015). Water quality and quantity impacts of hydraulic fracturing. Current Sustainable/Renewable Energy Reports 2: 17-24. http://dx.doi.org/10.1007/s40518-014-0023-4

LEau LLC. (2008). Dew vaporation desalination 5,000-gallon-per-day pilot plant. (Desalination and Water Purification Research and Development Program Report No. 120). Denver, CO: Bureau of Reclamation, U.S. Department of the Interior. http://www.usbr.gov/research/AWT/reportpdfs/report120.pdf

LeBas, R; Lord, P; Luna, D; Shahan, T. (2013). Development and use of high-TDS recycled produced water for crosslinked-gel-based hydraulic fracturing. In 2013 SPE hydraulic fracturing technology conference. Richardson, TX: Society of Petroleum Engineers. http://dx.doi.org/10.2118/163824-MS

Lefebvre, O; Moletta, R. (2006). Treatment of organic pollution in industrial saline wastewater: a literature review [Review]. Water Res 40: 3671-3682. http://dx.doi.org/10.1016/j.watres.2006.08.027

Linarić, M; Markić, M; Sipos, L. (2013). High salinity wastewater treatment. Water Sci Technol 68: 1400-1405. http://dx.doi.org/10.2166/wst.2013.376

Lutz, BD; Lewis, AN; Doyle, MW. (2013). Generation, transport, and disposal of wastewater associated with Marcellus Shale gas development. Water Resour Res 49: 647-656.

Ma, G; Geza, M; Xu, P. (2014). Review of flowback and produced water management, treatment, and beneficial use for major shale gas development basins. Shale Energy Engineering Conference 2014, Pittsburgh, Pennsylvania, United States.

Malone, S; Kelso, M; Auch, T; Edelstein, K; Ferrar, K; Jalbert, K. (2015). Data inconsistencies from states with unconventional oil and gas activity. J Environ Sci Health A Tox Hazard Subst Environ Eng 50: 501-510. http://dx.doi.org/10.1080/10934529.2015.992678

Maloney, KO; Yoxtheimer, DA. (2012). Production and disposal of waste materials from gas and oil extraction from the Marcellus shale play in Pennsylvania. Environmental Practice 14: 278-287. http://dx.doi.org/10.10170S146604661200035X

Mantell, ME. (2013a). Recycling and reuse of produced water to reduce freshwater use in hydraulic fracturing operations. Presentation presented at Water acquisition modeling: Assessing impacts through modeling and other means Technical workshop on wastewater treatment and related modeling, June 4, 2013, Arlington, VA.

Matthews, JC; Li, S; Swann, CT; Ericksen, RL. (2006). Incubation with moist top soils enhances solubilization of radium and other components from oilfield scale and sludge: Environmental concerns from Mississippi. Environmental Geosciences 13: 43-53.

Minnich, K. (2011). A water chemistry perspective on flowback reuse with several case studies. Minnich, K. http://www2.epa.gov/sites/production/files/documents/10_Minnich_-_Chemistry_508.pdf

Morillon, A; Vidalie, JF; Syahnudi, U; Suripno, S; Hadinoto, EK. (2002). Drilling and waste management; SPE 73931. Presentation presented at The SPE International Conference on Health, Safety, and Environment in Oil and Gas Exploration and Production, March 20-22, 2002, Kuala Lumpur, Malaysia.

Murray, KE. (2013). State-scale perspective on water use and production associated with oil and gas operations, Oklahoma, U.S. Environ Sci Technol 47: 4918-4925. http://dx.doi.org/10.1021/es4000593

Nelson, AW; May, D; Knight, AW; Eitrheim, ES; Mehrhoff, M; Shannon, R; Litman, R; Schultz, MK. (2014). Matrix complications in the determination of radium levels in hydraulic fracturing flowback water from Marcellus Shale. 1: 204-208. http://dx.doi.org/10.1021/ez5000379

NETL (National Energy Technology Laboratory). (2014). Evaporation [Fact Sheet]. Pittsburgh, PA: US Department of Energy. http://www.netl.doe.gov/research/coal/crosscutting/pwmis/tech-desc/evap

Nicot, JP; Reedy, RC; Costley, RA; Huang, Y. (2012). Oil & gas water use in Texas: Update to the 2011 mining water use report. Nicot, JP; Reedy, RC; Costley, RA; Huang, Y. http://www.twdb.state.tx.us/publications/reports/contracted_reports/doc/0904830939_2012Update_MiningWaterUse.pdf

Nicot, JP; Scanlon, BR. (2012). Water use for shale-gas production in Texas, U.S. Environ Sci Technol 46: 3580-3586. http://dx.doi.org/10.1021/es204602t

Nowak, N; Bradish, J. (2010). High density polyethylene (HDPE) lined produced water evaporation ponds. Presentation presented at 17th International Petroleum and Biofuels Environmental Conference, August 31-September 2, 2010, San Antonio, TX.

NPC (National Petroleum Council). (2011a). Management of produced water from oil and gas wells. (Paper #2-17). Washington, DC. http://www.npc.org/Prudent_Development-Topic_Papers/2-17_Management_of_Produced_Water_Paper.pdf

NRC (National Research Council). (2010). Management and effects of coalbed methane produced water in the western United States. Washington, DC: National Academies Press. http://www.nap.edu/catalog.php?record_id=12915

Olmstead, SM; Muehlenbachs, LA; Shih, JS; Chu, Z; Krupnick, AJ. (2013). Shale gas development impacts on surface water quality in Pennsylvania. PNAS 110: 4962-4967. http://dx.doi.org/10.1073/pnas.1213871110

Olsson, O; Weichgrebe, D; Rosenwinkel, KH. (2013). Hydraulic fracturing wastewater in Germany: composition, treatment, concerns. Environmental Earth Sciences 70: 3895-3906. http://dx.doi.org/10.1007/s12665-013-2535-4

Orem, WH; Tatu, CA; Lerch, HE; Rice, CA; Bartos, TT; Bates, AL; Tewalt, S; Corum, MD. (2007). Organic compounds in produced waters from coalbed natural gas wells in the Powder River Basin, Wyoming, USA. Appl Geochem 22: 2240-2256. http://dx.doi.org/10.1016/j.apgeochem.2007.04.010

PA DEP (Pennsylvania Department of Environmental Protection). (2011a). Letter from Pennsylvania Department of the Environment to US EPA Region 3 Administrator Shawn Garvin. Available online at http://www.epa.gov/region3/marcellus_shale?#inforeqsbypadep

PA DEP. Road-spreading of brine for dust control and road stabilization, 43 Pa.B. 7377 § 78.70 (2011b). http://www.pabulletin.com/secure/data/vol43/43-50/2362b.html

PA DEP (Pennsylvania Department of Environmental Protection). (2015a). PA DEP oil & gas reporting website, statewide data downloads by reporting period. waste and production files downloaded for Marcellus/unconventional wells, July 2009 December 2014. Harrisburg, PA. Retrieved from https://www.paoilandgasreporting.state.pa.us/publicreports/Modules/DataExports/DataExports.aspx

PA DEP (Pennsylvania Department of Environmental Protection). (2015b). Technologically enhanced naturally occurring radioactive materials (TENORM) study report. Harrisburg, PA. http://www.elibrary.dep.state.pa.us/dsweb/Get/Document-105822/PA-DEP-TENORM-Study_Report_Rev._0_01-15-2015.pdf

Parker, KM; Zeng, T; Harkness, J; Vengosh, A; Mitch, WA. (2014). Enhanced formation of disinfection byproducts in shale gas wastewater-impacted drinking water supplies. Environ Sci Technol 48: 11161-11169. http://dx.doi.org/10.1021/es5028184

Pashin, JC; Mcintyre-Redden, MR; Mann, SD; Kopaska-Merkel, DC; Varonka, M; Orem, W. (2014). Relationships between water and gas chemistry in mature coalbed methane reservoirs of the Black Warrior Basin. Int J Coal Geol 126: 92-105. http://dx.doi.org/10.1016/j.coal.2013.10.002

Peraki, M; Ghazanfari, E. (2014). Electrodialysis treatment of flow-back water for environmental protection in shale gas development. In Shale gas development Shale energy engineering 2014. Reston, VA: American Society of Civil Engineers. http://dx.doi.org/10.1061/9780784413654.008

Plumlee, MH; Debroux, JF; Taffler, D; Graydon, JW; Mayer, X; Dahm, KG; Hancock, NT; Guerra, KL; Xu, P; Drewes, JE; Cath, TY. (2014). Coalbed methane produced water screening tool for treatment technology and beneficial use. 5: 22-34. http://dx.doi.org/10.1016/j.juogr.2013.12.002

Porcelli, D; Kim, CK; Martin, P; Moore, WS; Phaneuf, M. (2014). Properties of radium. In The environmental behaviour of radium: revised edition. Vienna, Austria: International Atomic Energy Agency. http://www-pub.iaea.org/MTCD/Publications/PDF/trs476_web.pdf

Purestream (Purestream Services). (2013). Purestream services will begin commercial operations to treat Eagle Ford Shale produced and frac flow-back water in Gonzalez County, Texas. Retrieved from http://purestreamtechnology.com/index.php/component/content/article/72-press-releases/206-purestream-services-will-begin-commercial-operations-to-treat-eagle-ford-shale-produced-and-frac-flow-back-water-in-gonzalez-county-texas

Rahm, BG; Bates, JT; Bertoia, LR; Galford, AE; Yoxtheimer, DA; Riha, SJ. (2013). Wastewater management and Marcellus Shale gas development: trends, drivers, and planning implications. J Environ Manage 120: 105-113. http://dx.doi.org/10.1016/j.jenvman.2013.02.029

Rahm, BG; Riha, SJ. (2014). Evolving shale gas management: water resource risks, impacts, and lessons learned [Review]. Environ Sci Process Impacts 16: 1400-1412. http://dx.doi.org/10.1039/c4em00018h

Ramirez, P. (2002). Oil field produced water discharges into wetlands in Wyoming. (97-6- 6F34). Cheyenne, WY: U.S. Fish and Wildlife Service. http://www.fws.gov/mountain-prairie/contaminants/papers/r6718c02.pdf

Rassenfoss, S. (2011). From flowback to fracturing: Water recycling grows in the Marcellus shale. J Pet Tech 63: 48-51.

Resnikoff, M; Alexandrova, E; Travers, J. (2010). Radioactivity in Marcellus shale: Report prepared for Residents for the Preservation of Lowman and Chemung (RFPLC). New York, NY: Radioactive Waste Management Associates.

Rich, AL; Crosby, EC. (2013). Analysis of reserve pit sludge from unconventional natural gas hydraulic fracturing and drilling operations for the presence of technologically enhanced naturally occurring radioactive material (TENORM). New Solutions: A Journal of Environmental and Occupational Health Policy 23: 117-135. http://dx.doi.org/10.2190/NS.23.1.h

Richardson, SD; Plewa, MJ; Wagner, ED; Schoeny, R; Demarini, DM. (2007). Occurrence, genotoxicity, and carcinogenicity of regulated and emerging disinfection by-products in drinking water: A review and roadmap for research [Review]. Mutat Res 636: 178-242. http://dx.doi.org/10.1016/j.mrrev.2007.09.00

Rowan, EL; Engle, MA; Kirby, CS; Kraemer, TF. (2011). Radium content of oil- and gas-field produced waters in the northern Appalachian Basin (USA): Summary and discussion of data. (Scientific Investigations Report 20115135). Reston, VA: U.S. Geological Survey. http://pubs.usgs.gov/sir/2011/5135/

Rozell, DJ; Reaven, SJ. (2012). Water pollution risk associated with natural gas extraction from the Marcellus Shale. Risk Anal 32: 13821393. http://dx.doi.org/10.1111/j.1539-6924.2011.01757.x

Rushton, L; Castaneda, C. (2014). Drilling into hydraulic fracturing and the associated wastewater management issues. Washington, WD: Paul Hastings, LLP. http://www.paulhastings.com/docs/default-source/PDFs/stay-current-hydraulic-fracturing-wastewater-management.pdf

Schmidt, CW. (2013). Estimating wastewater impacts from fracking. Environ Health Perspect 121: A117. http://dx.doi.org/10.1289/ehp.121-a117

Schubert, J; Rosenmeier, J; Zatezalo, M. (2014). A review of NORM/TENORM in wastes and waters associated with Marcellus shale gas development and production. In CL Meehan; JM Vanbriesen; F Vahedifard; X Yu; C Quiroga (Eds.), Shale energy engineering 2014: technical challenges, environmental issues, and public policy (pp. 492-501). Reston, VA: American Society of Civil Engineers. http://dx.doi.org/10.1061/9780784413654.052

Shafer, L. (2011). Water recycling and purification in the Pinedale anticline field: results from the anticline disposal project. In 2011 SPE Americas E&P health, safety, security & environmental conference. Richardson, TX: Society of Petroleum Engineers. http://dx.doi.org/10.2118/141448-MS

Shaffer, DL; Arias Chavez, LH; Ben-Sasson, M; Romero-Vargas Castrillón, S; Yip, NY; Elimelech, M. (2013). Desalination and reuse of high-salinity shale gas produced water: drivers, technologies, and future directions. Environ Sci Technol 47: 9569-9583.

Silva, JM; Matis, H; Kostedt, WL; Watkins, V. (2012). Produced water pretreatment for water recovery and salt production. (08122-36). Niskayuna, NY: Research Partnership to Secure Energy for America. http://www.rpsea.org/media/files/project/18621900/08122-36-FR-Pretreatment_Water_Mgt_Frac_Water_Reuse_Salt-01-26-12.pdf

Sionix (Sionix Corporation). (2011). Sionix to build Bakken water treatment plant. Retrieved from http://www.rigzone.com/news/article_pf.asp?a_id=110613

Sirivedhin, T; Dallbauman, L. (2004). Organic matrix in produced water from the Osage-Skiatook petroleum environmental research site, Osage county, Oklahoma. Chemosphere 57: 463-469.

Skalak, KJ; Engle, MA; Rowan, EL; Jolly, GD; Conko, KM; Benthem, AJ; Kraemer, TF. (2014). Surface disposal of produced waters in western and southwestern Pennsylvania: Potential for accumulation of alkali-earth elements in sediments. Int J Coal Geol 126: 162-170. http://dx.doi.org/10.1016/j.coal.2013.12.001

Slutz, J; Anderson, J; Broderick, R; Horner, P. (2012). Key shale gas water management strategies: An economic assessment tool. Paper presented at International Conference on Health, Safety and Environment in Oil and Gas Exploration and Production, September 11-13, 2012, Perth, Australia.

Soeder, DJ; Kappel, WM. (2009). Water resources and natural gas production from the Marcellus shale [Fact Sheet] (pp. 6). (U.S. Geological Survey, Fact Sheet 20093032). Soeder, DJ; Kappel, WM. http://pubs.usgs.gov/fs/2009/3032/pdf/FS2009-3032.pdf

States, S; Cyprych, G; Stoner, M; Wydra, F; Kuchta, J; Monnell, J; Casson, L. (2013). Marcellus Shale drilling and brominated THMs in Pittsburgh, Pa., drinking water. J Am Water Works Assoc 105: E432-E448. http://dx.doi.org/10.5942/jawwa.2013.105.0093

Stewart, DR. (2013b). Treatment for beneficial use of produced water and hydraulic fracturing flowback water. Presentation presented at US EPA Technical Workshop on Wastewater Treatment and Related Modeling For Hydraulic Fracturing, April 18, 2013, Research Triangle Park, NC.

Sumi, L. (2004). Pit pollution: Backgrounder on the issues, with a New Mexico case study. Washington, DC: Earthworks: Oil and Gas Accountability Project. http://www.earthworksaction.org/files/publications/PitReport.pdf

Swann, C; Matthews, J; Ericksen, R; Kuszaul, J. (2004). Evaluations of radionuclides of uranium, thorium, and radium with produced fluids, precipitates, and sludges from oil, gas, and oilfield brine injections wells. (DE-FG26-02NT 15227). Washington, D.C.: U.S. Department of Energy. http://www.olemiss.edu/depts/mmri/programs/norm_final.pdf

Tiemann, M; Folger, P; Carter, NT. (2014). Shale energy technology assessment: Current and emerging water practices. Washington, DC: Congressional Research Service. http://nationalaglawcenter.org/wp-content/uploads//assets/crs/R43635.pdf

Titler, RV; Curry, P. (2011). Chemical analysis of major constituents and trace contaminants of rock salt. Harrisburg, PA: Pennsylvania Department of Environmental Protection. http://files.dep.state.pa.us/water/Wastewater%20Management/WastewaterPortalFiles/Rock%20Salt%20Paper%20final%20052711.pdf

U.S. Department of Justice. (2014). Company owner sentenced to more than two years in prison for dumping fracking waste in Mahoning River tributary. Available online at http://www.justice.gov/usao/ohn/news/2014/05auglupo.html (accessed March 4, 2015).

U.S. EPA (U.S. Environmental Protection Agency). (2006). National Primary Drinking Water Regulations: Stage 2 Disinfectants and Disinfection Byproducts Rule. http://water.epa.gov/lawsregs/rulesregs/sdwa/stage2/

U.S. EPA (U.S. Environmental Protection Agency). (2014f). Minimizing and managing potential impacts of injection-induced seismicity from class II disposal wells: Practical approaches [EPA Report]. Washington, D.C. http://www.epa.gov/r5water/uic/ntwg/pdfs/induced-seismicity-201502.pdf

U.S. EPA (U.S. Environmental Protection Agency). (2015d). DMR spreadsheet Pennsylvania wastewater treatment plants per Region 3 Information Request. Data provided by request. Washington , D.C.: Region 3, U.S. Environmental Protection Agency.

U.S. EPA (U.S. Environmental Protection Agency). (2015e). Effluent data from Pennsylvania wastewater treatment plants per Region 3 Information Request. Data provided by request. Washington, D.C.: Region 3, U.S. Environmental Protection Agency.

U.S. EPA (U.S. Environmental Protection Agency). (2015f). EPA Enforcement and Compliance History. Online: Effluent Charts: SEECO-Judsonia Water Reuse Recycling Facility. Available online at http://echo.epa.gov/effluent-charts#AR0052051

U.S. EPA (U.S. Environmental Protection Agency). (2015h). Key documents about mid-Atlantic oil and gas extraction. Available online at http://www.epa.gov/region3/marcellus_shale/#aoinfoww (accessed May 7, 2015).

U.S. EPA. National primary drinking water regulations public notification rule and consumer confidence report rule health effects language. (parts 141.201, and 141.151), (U.S. Government Publishing Office2015i). http://www.ecfr.gov/cgi-bin/text-idx?SID=4d25ec04bc44e54b1efdf307855f3185&node=pt40.23.141&rgn=div5

U.S. EPA (U.S. Environmental Protection Agency). (2015p). Sources contributing bromide and inorganic species to drinking water intakes on the Allegheny river in western Pennsylvania [EPA Report]. (EPA/600/R-14/430). Washington, D.C.

U.S. EPA (U.S. Environmental Protection Agency). (2015q). Technical development document for proposed effluent limitation guidelines and standards for oil and gas extraction. (EPA-821-R-15-003). Washington, D.C. http://water.epa.gov/scitech/wastetech/guide/oilandgas/unconv.cfm

U.S. GAO (U.S. Government Accountability Office). (2012). Energy-water nexus: Information on the quantity, quality, and management of water produced during oil and gas production. (GAO-12-156). Washington, D.C. http://www.gao.gov/products/GAO-12-156

USGS (U.S. Geological Survey). (2013c). National Water Information System (NWIS) [Database]. Retrieved from http://waterdata.usgs.gov/nwis

USGS (U.S. Geological Survey). (2014e). U.S. Geological Survey national produced waters geochemical database v2.0 (PROVISIONAL). Available online at http://energy.usgs.gov/EnvironmentalAspects/EnvironmentalAspectsofEnergyProductionandUse/ProducedWaters.aspx#3822349-data

Van Voast, WA. (2003). Geochemical signature of formation waters associated with coalbed methane. AAPG Bulletin 87: 667-676.

Veil, JA. (2011). Water management practices used by Fayetteville shale gas producers. (ANL/EVS/R-11/5). Washington, DC: U.S. Department of Energy, National Energy Technology Laboratory. http://www.ipd.anl.gov/anlpubs/2011/06/70192.pdf

Veil, JA; Puder, MG; Elcock, D; Redweik, RJ. (2004). A white paper describing produced water from production of crude oil, natural gas, and coalbed methane. Lemont, IL: Argonne National Laboratory.

Vengosh, A; Jackson, RB; Warner, N; Darrah, TH; Kondash, A. (2014). A critical review of the risks to water resources from unconventional shale gas development and hydraulic fracturing in the United States. Environ Sci Technol 48: 36-52. http://dx.doi.org/10.1021/es405118y

Vidic, RD; Brantley, SL; Vandenbossche, JM; Yoxtheimer, D; Abad, JD. (2013). Impact of shale gas development on regional water quality [Review]. Science 340: 1235009. http://dx.doi.org/10.1126/science.1235009

Volz, CD; Ferrar, K; Michanowicz, D; Christen, C; Kearney, S; Kelso, M; Malone, S. (2011). Contaminant Characterization of Effluent from Pennsylvania Brine Treatment Inc., Josephine Facility Being Released into Blacklick Creek, Indiana County, Pennsylvania: Implications for Disposal of Oil and Gas Flowback Fluids from Brine Treatment Plants [Standard]. Volz, CD; Ferrar, K; Michanowicz, D; Christen, C; Kearney, S; Kelso, M; Malone, S. http://www2.epa.gov/hfstudy/contaminant-characterization-effluent-pennsylvania-brine-treatment-inc-josephine-facility

Walsh, JM. (2013). Water management for hydraulic fracturing in unconventional resourcesPart 1. Oil and Gas Facilities 2.

Walter, GR; Benke, RR; Pickett, DA. (2012). Effect of biogas generation on radon emissions from landfills receiving radium-bearing waste from shale gas development. J Air Waste Manag Assoc 62: 1040-1049. http://dx.doi.org/10.1080/10962247.2012.696084

Warner, NR; Christie, CA; Jackson, RB; Vengosh, A. (2013a). Impacts of shale gas wastewater disposal on water quality in western Pennsylvania. Environ Sci Technol 47: 11849-11857. http://dx.doi.org/10.1021/es402165b

Warner, NR; Kresse, TM; Hays, PD; Down, A; Karr, JD; Jackson, RB; Vengosh, A. (2013b). Geochemical and isotopic variations in shallow groundwater in areas of the Fayetteville Shale development, north-central Arkansas. Appl Geochem 35: 207-220.

Water Research Foundation. (2010). Assessment of inorganics accumulation in drinking water system scales and sediments. Denver, CO. http://www.waterrf.org/PublicReportLibrary/3118.pdf

Weaver, JW; Xu, J; Mravik, SC. (In Press) Scenario analysis of the impact on drinking water intakes from bromide in the discharge of treated oil and gas waste water. J Environ Eng.

Webb, CH; Nagghappan, L; Smart, G; Hoblitzell, J; Franks, R. (2009). Desalination of oilfield-produced water at the San Ardo water reclamation facility, Ca. In SPE Western regional meeting 2009. Richardson, TX: Society of Petroleum Engineers. http://dx.doi.org/10.2118/121520-MS

Wendel, K. (2011). Wastewater technologies critical for continued growth of Marcellus. Available online at http://www.ogfj.com/articles/print/volume-8/issue-11/features/wastewater-technologies-critical-for.html (accessed March 9, 2015).

Wess, J; Ahlers, H; Dobson, S. (1998). Concise International Chemical Assessment Document 10: 2-Butoxyethanol. World Health Organization. http://www.who.int/ipcs/publications/cicad/cicad_10_revised.pdf

West Virginia DEP (West Virginia Department of Environmental Protection). (2011). Memorandum of agreement from the Division of Water and Waste Management to the Division of Highway: WVDOH/WVDEP Salt brine from gas wells agreement. Available online at http://www.dep.wv.gov/WWE/Documents/WVDOHWVDEP%20Salt%20Brine%20Agreement.pdf

Wilson, JM; Van Briesen, JM. (2013). Source water changes and energy extraction activities in the Monongahela River, 2009-2012. Environ Sci Technol 47: 1257512582. http://dx.doi.org/10.1021/es402437n

Wilson, JM; Vanbriesen, JM. (2012). Oil and gas produced water management and surface drinking water sources in Pennsylvania. Environmental Practice 14: 288-300.

Wolfe, D; Graham, G. (2002). Water rights and beneficial use of produced water in Colorado. Denver, CO: American Water Resources Association. http://www.gwpc.org/sites/default/files/event-sessions/Dick_Wolfe_PWC02_0.pdf

Xu, P; Drewes, JE; Heil, D. (2008). Beneficial use of co-produced water through membrane treatment: Technical-economic assessment. Desalination 225: 139-155. http://dx.doi.org/10.1016/j.desal.2007.04.093

Younos, T; Tulou, KE. (2005). Overview of desalination techniques. Journal of Contemporary Water Research & Education 132: 3-10. http://dx.doi.org/10.1111/j.1936-704X.2005.mp132001002.x

Zhang, T; Gregory, K; Hammack, RW; Vidic, RD. (2014b). Co-precipitation of radium with barium and strontium sulfate and its impact on the fate of radium during treatment of produced water from unconventional gas extraction. Environ Sci Technol 48: 4596-4603. http://dx.doi.org/10.1021/es405168b